Quantitative Health Risk Assessment of Recreational Water Users in Philadelphia

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DEDICATIONS

This dissertation is dedicated to my loving mother and late father for their endless love and stupendous faith in my aspirations. Their perseverance and hard work have encouraged me throughout my life.

A special feeling of gratitude for my immensely supportive and super brilliant husband, Gautam Malhotra, our adventurous, strong willed and precious little girl Arsha Malhotra and for my outstandingly amazing in-laws for their constant support and sacrifices towards making me achieve my goals.

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ABSTRACT

Quantitative health risk assessment of recreational water users in Philadelphia
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Philadelphia's water resources are widely used for a range of recreational activities and since almost 60% of the City is served by Combined Sewer Systems (CSS), the pathogen levels after wet weather discharges are of main concern for evaluating the hygienic status of City's these resources. A quantitative microbial risk assessment (QMRA) was conducted to estimate risk of gastrointestinal illnesses (GI) associated with recreational exposures to Philadelphia water bodies, under dry and wet weather conditions. Usage pattern was measured at ten different locations (creek sites: n=3 and river sites: n=7) by using a novel ''time lapse photography'' technology during three recreational seasons (May-September) 2008-2010. Using maximum likelihood estimation stochastic exposure models were generated for each exposure scenario. In general, the log normal distribution described the playing and wading duration distribution, while the gamma distribution was the best fit for fishing durations.

Both *E. coli* and *Enterococcus* were evaluated as water quality indicators and daily risk of GI illnesses in proportion of the frequency of use, including all types of activities that span the range of exposure at each site, was predicted by running 10,000 Monte Carlo simulations in Crystal Ball. An exponential dose response model was developed based on the 1986 US EPA epidemiological studies and the results were compared with the EPA's 1986-criteria model. A base line approach was developed to extend the risk estimates from local-observation stations to the entire stretch of the water bodies and critical zones of the Schuylkill and the Delaware River were identified for further investigations.

Consistently, the sites on small creeks presented elevated risk potential under episodic events of poor water quality. Activities resulting in greatest number of affected users at creeks were identified as wading and playing (81%), while fishing was the potential risk contributor (65%) at rivers. To our knowledge this is the first QMRA to employ time-lapse cameras to characterize exposure types and durations. The quantitative measure of risk contribution from each type of water activity and identification of critical zones on large water bodies can be useful for policy makers in planning the health campaigns and prioritizing the future interventions.

1 Chapter 1: INTRODUCTION

Water is an essential component of the earth's ecosystem. Uses of water include agricultural, industrial, household, recreational and environmental activities. These uses can be further classified as withdrawal and non-withdrawal uses. Basically, the withdrawal type of uses includes the activities that remove water from the ground or divert it from a surface-source for use; however the non-withdrawal uses do not require any diversions (MacKichan 1961; Kenny, Barber et al. 2009). The common practices classified as non-withdrawal water use are navigation, recreation and conservation of wildlife. Out of these, recreation constitutes a very small percentage of the total water use, but is becoming more and more important in context of public health assessment because people are reported to have more leisure time than formerly (Ausubel and Grubler 1995; Roberts Kenneth 2006).

Relating recreational water exposure to health outcomes is a complicated research topic because both the water usage and bacterial loads vary temporally and spatially across a watershed. These variations are primarily related to rainfall occurrences. Rains of sufficient intensity may cause Combined Sewer Overflows (CSOs), resulting in discharge of raw sewage into fresh water bodies. Any recreational exposure to this water, during periods of increased pathogen concentration may lead to significantly higher risk of illness.

In order to ensure the health of the users and to protect the water resources from deterioration, there are directed regulations administered by federal and state governments. The Clean Water Act (CWA, 1972) is one of the most successful environmental regulations of US EPA which established a basic framework to regulate discharges of pollutants into waters. Under the CWA, water quality standards were developed to define the goals for a water body by designating its uses and by setting criteria to protect those uses (U.S. EPA 1972). Though these water quality criteria sets are generally protective of the nation's waters, they do not reflect the specific risk of illness posed by recreational use of a particular water body. There is a vast recognition that the current recreational water standards do not adequately account for different usages of recreational waters, and that with new detection methods it is possible to test additional microbial agents more rapidly.

The bacteriological criteria developed by US EPA for recreational water protection are based on the acceptable risk levels for fresh and marine waters, providing compliance with the "swimmable" goal of CWA. It relates excess HCGI illnesses (Highly Credible Gastrointestinal Illness) to concentrations of fecal indicators in recreational waters and recommends regulatory standards for the bacteria to limit the contamination of waters (U.S. EPA 1986; U.S. EPA 2004). This approach has two major limitations; it does not specify any quantitative measure of actual health risk from recreation in the waterways and it does not distinguish between different recreational uses such as boating, fishing, wading and playing. Since non-swimming uses are predominant in Philadelphia waters and nationwide, many cities, particularly those with combined sewers, are making massive investments for "gray" infrastructure to reduce storm water runoff; the Philadelphia Water Department (PWD) initiated this study by Drexel to investigate an alternative approach to quantify the influence of the wet weather discharges on the recreational user's health. Specifically, the aim of the study is to evaluate the risk of gastrointestinal illnesses (GI) among the users involved in various types of recreational activities under the wet weather influences and to project how the risk potential varies across the entire water body to help the watershed managers in allocating their resources more efficiently.

1.1 Background

1.1.1. Characterization of Philadelphia Rivers and Creeks

The City of Philadelphia has two major rivers -- the Schuylkill and the Delaware River and has a total area of 142.6 square miles. It was developed by William Penn, an English real estate entrepreneur and philosopher who envisioned it as a *Green Country Towne* over three centuries ago (1682). However as the city grew into bustling industrial hub, it faced the rapid decline of its sanitary system. As a common practice the industrial and human waste was dumped into creeks and streams, which finally discharged into the two large rivers. This water pollution continued until there were frequent and widespread epidemics (1880's and 1890's), which lead the city officials to develop an infrastructure networks of combine sewers placing conveyance pipes in creeks' beds. But the sewage was still being dumped directly into the Schuylkill and Delaware Rivers, polluting the City's water supply. Consequently, in 1905 the Pennsylvania legislature enacted the Clean Streams Law which after major amendments, in 1970 mandated the end to sewage pollution of the Pennsylvania's streams (Smith 1990). As a solution to increasing river pollution, the city built a network of interceptor sewers dividing the wastewater and stormwater conveyance system into three drainage districts: Northeast, Southeast and Southwest. The sewage from each district was diverted to its respective wastewater treatment plant before it was discharged to the rivers.

The transformation of streams carrying sewage into huge pipes changed the city's topography and hydrology drastically. Almost 73% of the historic waterways were converted into sewers leaving only 118 linear miles of streams in the city (Levine 2010). But the commitment of the city to protect its people from water crisis, lead the city to become the first American city during 1801 to take the responsibility of supplying public drinking water to its residents (Kramek and Loh 2007). The city pioneered the centralized water distribution system by setting up three drinking water treatment plants (the Queen Lane treatment plant and the Belmont water treatment plant on the Schuylkill River; The Baxter water treatment plant on the Delaware River). The city officials emphasized the connection between clean water and parklands; and with an aim to promote healthy communities they purchased land along the Schuylkill River in 1867 and created a massive green buffer area known as Fairmount Park, now the world's largest urban park (Klein and Fairmount Park Commission. Philadelphia 1974; McBride, Cianfrani et al. 2002; Kramek and Loh 2007).

The entire city drainage area is divided into seven watersheds (Figure1.1). These are (from north to south) Poquessing Creek watershed, Pennypack Creek watershed, Delaware Direct watershed, Tacony and Frankford Creeks watershed, Wissahickon Creek watershed , Lower Schuylkill River watershed, and Cobbs Creek watershed. Four of these seven watersheds, generally comprised of older areas of the city, receive CSO discharges (Table 1.1). The following section of the thesis will discuss the physical characteristics and management approaches for each of these 4 watersheds in detail.

Table 1-1 Watershed areas served by CSO discharges.

Watersheds Receiving CSO discharges	Area drained within Philadelphia (sq. miles)	Percent area served by CSO (%, approx.)
Schuylkill River	36	40
Delaware River	40	71
Cobbs Creek	6	80
Tacony-Frankford Creek	19	80

Source: PWD, 2011

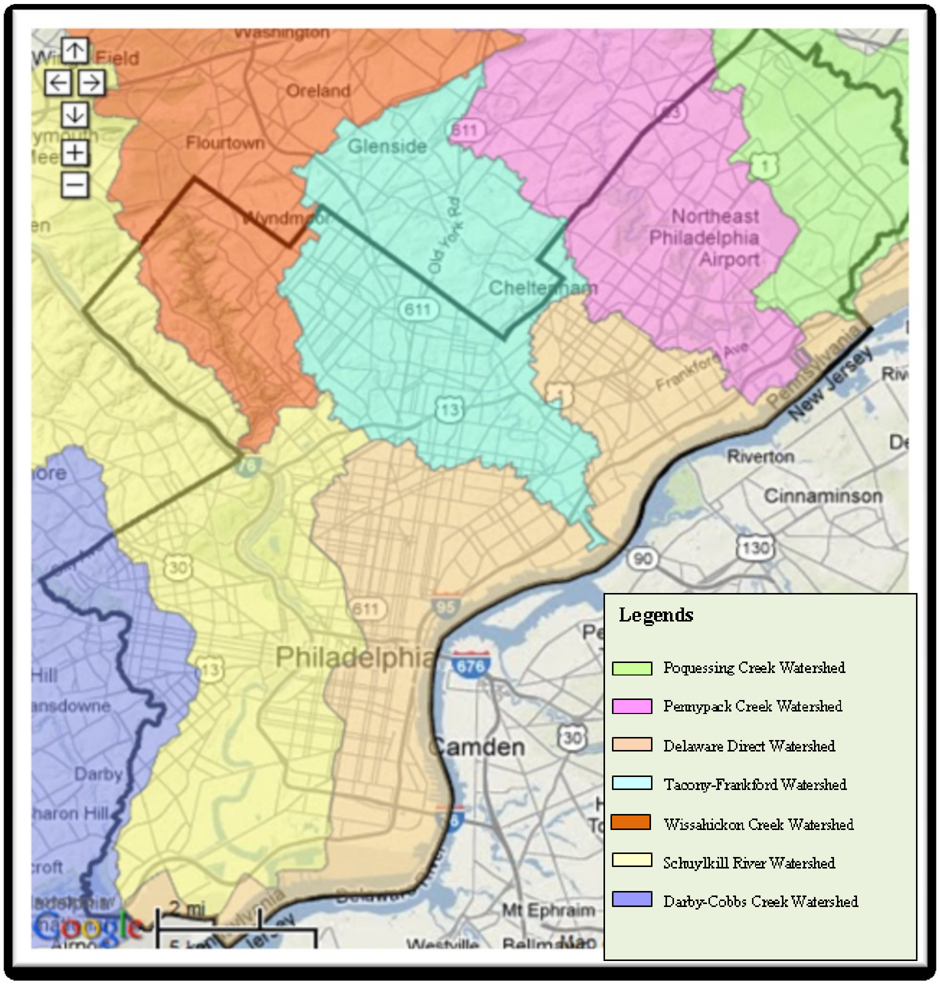

Figure 1-1 Watershed boundaries within the City of Philadelphia

1.1.1.1 Lower Schuylkill River watershed: The Schuylkill River is the largest tributary to the Delaware River. It also plays an important role as a fish habitat and source of drinking water. The entire watershed of the Schuylkill River covers approximately 1916 square miles but only

2% of this area falls under Philadelphia County. The four major counties covered in large parts under this watershed are- Berks, Montgomery, Schuylkill, and Chester counties (Fig. 1.2). The River travels approximately 130 miles from its headwaters at Tuscarora Springs in Schuylkill County, to its mouth at the Delaware River in Philadelphia (DCNR 2001). The major tributary of Schuylkill in Philadelphia is Wissahickon Creek with a watershed coverage area of approximately 64 square miles.

Two of the three drinking water treatment plants supplying water to the population of Philadelphia are located at the bottom of the Schuylkill River watershed. The Queen Lane intake is located 12 miles from the mouth of the Schuylkill River, directly downstream of the confluence of the Wissahickon Creek with the Schuylkill River. And the Belmont intake is located at approximately 2 miles upstream of the Fairmount Dam.

The management of this watershed falls under jurisdiction of all the counties covered by the watershed, but the advancements and the contributions made by PWD has been notable. In 1999 for protecting and improving the water supply to the Philadelphia residents, PWD established a Source Water Protection Program (SWPP) in which it identified the entire Schuylkill River watershed as its source water and initiated the development of source water protection plan. Between 1999 and 2003, PWD participated in the PADEP source water assessments as the primary contractor for surface water supplies in the Schuylkill River watershed. These assessments resulted in the identification of pollution sources posing the biggest threat to drinking water intakes along the Schuylkill. The priority contamination threats identified as a result of this program includes stormwater runoff, inappropriate agricultural practices and abandoned mine drainage. Other threats include sewage overflows and unplanned spills from industries (PWD 2006). In spring of 2003, as a transition from assessment to protection PWD in collaboration with US Environmental Protection Agency (EPA) Region 3 and PA Department of Environmental Protection (DEP) established the Schuylkill Action Network (SAN) workforce group which works in partnership with state agencies, local watershed organizations, water suppliers, local and federal governments to implement the protection and outreach projects. It is now established as a permanent watershed-wide organization, which is committed to identify problems, prioritize projects and fund sources to improve and restore the water quality throughout the Schuylkill River watershed (PWD and PDE 2004).

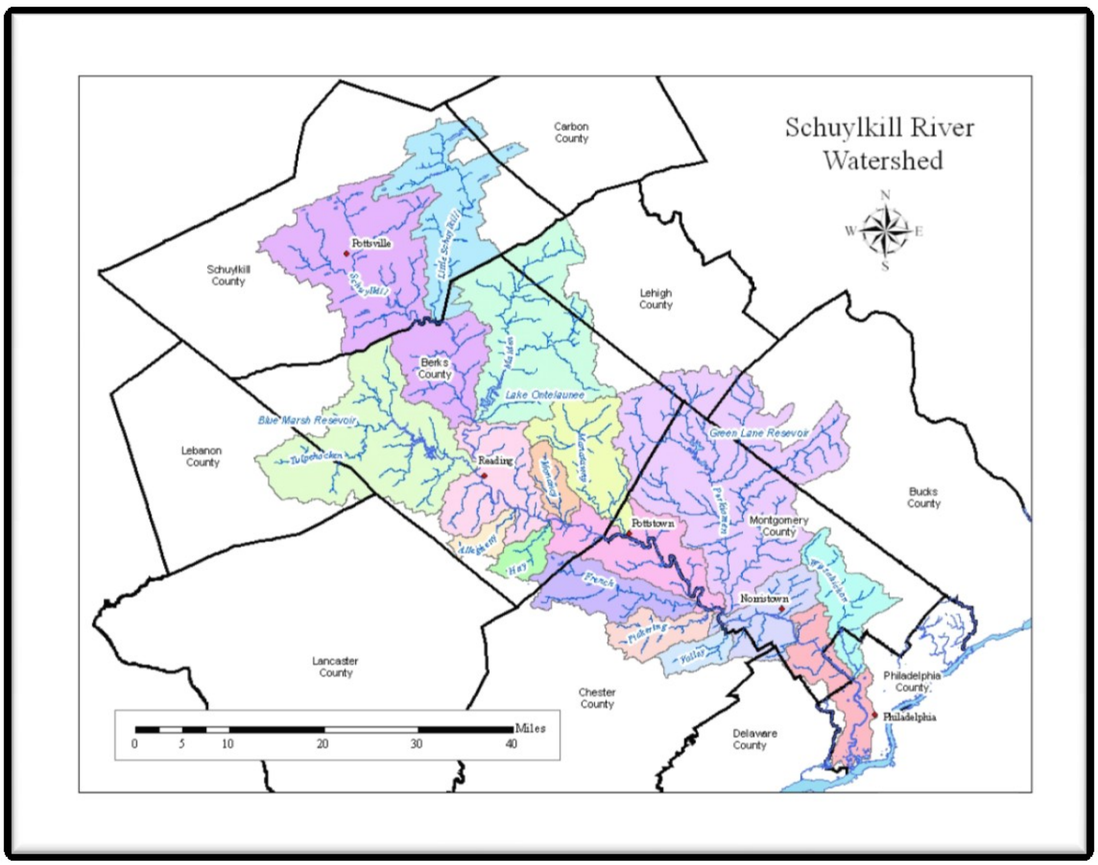

Figure 1-2 Schuylkill River drainage basin (Source: Schuylkill River Source Water Protection Plan-2006)

Another landmark in the efforts for the protection of this watershed is the development of the Schuylkill Watershed Conservation Plan during 2001 through a partnership between The Conservation Fund (TCF), Natural Lands Trust (NLT) and the Patrick Center for Environmental Research at the Academy of Natural Sciences (PCER - ANS). The project funding was obtained from the Pennsylvania Department of Conservation and Natural Resources (DCNR) - "Rivers Conservation Program in 1997 with a matching grant from The William Penn Foundation. This plan acts as a guidebook for municipalities, conservation groups, and for citizens interested in taking steps to enhance the long-term health of the Schuylkill River watershed (PWD 2006). There are other numerous conservation and education projects underway in this watershed, the repository of the information can be found at PA DCNR rivers website (http://www.dcnr.state.pa.us/rivers) and at Schuylkill Waters Watershed resources website (http://www.schuylkillwaters.org/watershed_resources.cfm).

Watershed Population and Land use types: Since the ownership of the Philadelphia land by William Penn in 1682, the population in Pennsylvania grew faster than the rest of the nation with a significant redistribution of people within the Philadelphia region. The headwater areas of the watershed in the Appalachian Mountain are lightly populated; the middle region expands over a gentle slope suitable for agricultural purposes and is relatively more populated. As the watershed flows downstream, it becomes increasingly suburban and finally gets heavily urbanized as it reaches Philadelphia. The most recent available census data by counties for Schuylkill river watershed is for the year 2010 and is summarized in Table 1.2(DCNR 2001).

Table 1-2 Census data for Schuylkill Watershed (Source: RCP report 2001)

County	2010 Population	2000 Population	1990 Population	1980 Population	% Change from 1980 - 2010	Population within Watershed
Berks	411,442	373,638	336,523	312,509	31.66	390,567
Bucks	625,249	597,635	541,174	479,211	30.47	48,519
Chester	498,886	433,501	376,396	316,660	57.55	122,132
Lebanon	133,568	120,327	113,744	108,582	23.01	17,952
Lehigh	349,497	312,090	291,130	272,349	28.33	10,493
Montgomery	799,874	750,097	678,111	643,621	24.28	575,834
Philadelphia	1,526,006	1,517,550	1,585,577	1,688,609	-9.63	398,519
Schuylkill	148,289	150,336	152,585	160,630	-7.68	81,154

Except for Philadelphia and Schuylkill County, all other counties in the Schuylkill watershed have experienced a significant increase in population over past 3 decades. The maximum population growth since 1980 is observed in Chester County however the population in Philadelphia County observed an overall drop with a slight increase since 2000. The central part of the watershed observed the greatest population growth mainly due to the influence of decentralization to the suburbs and was also induced due to economical out-migration from the Schuylkill County.

The primary land uses of the Schuylkill watershed have changed since its evolution from agricultural to industrial and now primarily to urban land type within the Philadelphia region. In early times, the settlers around the Schuylkill River relied on agriculture however with the exploration of large natural resources in the watershed such as; iron ore and hardwood, the land use type shifted from agricultural to industrial use. This rapid industrialization was followed by a peak in coal industry during 1910's when vast coal sources were discovered in the northern headwaters. This discovery converted the role of the Schuylkill River from a multi-purpose water supply system to a primary mode of transportation supported with the Schuylkill Navigation System comprising 32 dams and 103 locks. Later on by early 20th century the railroad development took over the navigation system by providing competing means of transportation of coal from source field to Philadelphia (PWD 2006).

According to the PWD’s 2002 report on the state of the Schuylkill River watershed, nearly 85% of the entire watershed remains in agriculture and forest but the area under Philadelphia County was identified as significantly urbanized. With incessant programs and efforts of PWD to restore the watershed these numbers are believed to be changed in the direction to increase the pervious surface with in the city to control storm water runoffs to the river. According to the most recent statistics available at PWD website, the current impervious area of Schuylkill watershed within Philadelphia County is approximately 10%.

1.1.1.2 Delaware Direct Watershed: The entire Delaware River watershed drains nearly 13,000 square miles, of which Philadelphia's contribution is less than 1%. The watershed encompasses areas of four states, 42 counties and parts of 838 municipalities in the Mid-Atlantic region. There is approximately 23,700 linear miles of streams in the watershed, but Philadelphia region covers only 21 miles but it has the highest population density of any county within this watershed (PWD 2011). The main stem of the river is the longest free flowing (undammed) river in the eastern United States serving a variety of residential and industrial purposes such as fishing, transportation, recreation, power cooling and most importantly drinking water supply to almost 10% of the US population.

Approximately 40 square miles of the City land drains directly to the Delaware River. The Philadelphia Water Department is focused on the management of water resources within city’s region of the watershed, designated as the Delaware Direct Watershed (DDW). DDW comprises the area of the City of Philadelphia that drains directly to the Delaware River and generally consists of the Delaware River Waterfront and several city blocks of inland area (Fig. 3). Non point sources of pollution, CSO's and stormwater overflows are amongst the highest priority problems in DDW as identified by PWD. According to the recent Source Water Assessment report of PWD, the Tidal PA Philadelphia, NJ Tidal Lower and Tidal PA Bucks had the greatest number of dischargers per acre of drainage area. The dry weather discharges due to choked sewers, defective laterals and illicit cross connections have been reported to significantly contribute to the water quality concerns in this watershed (PWD 2007).

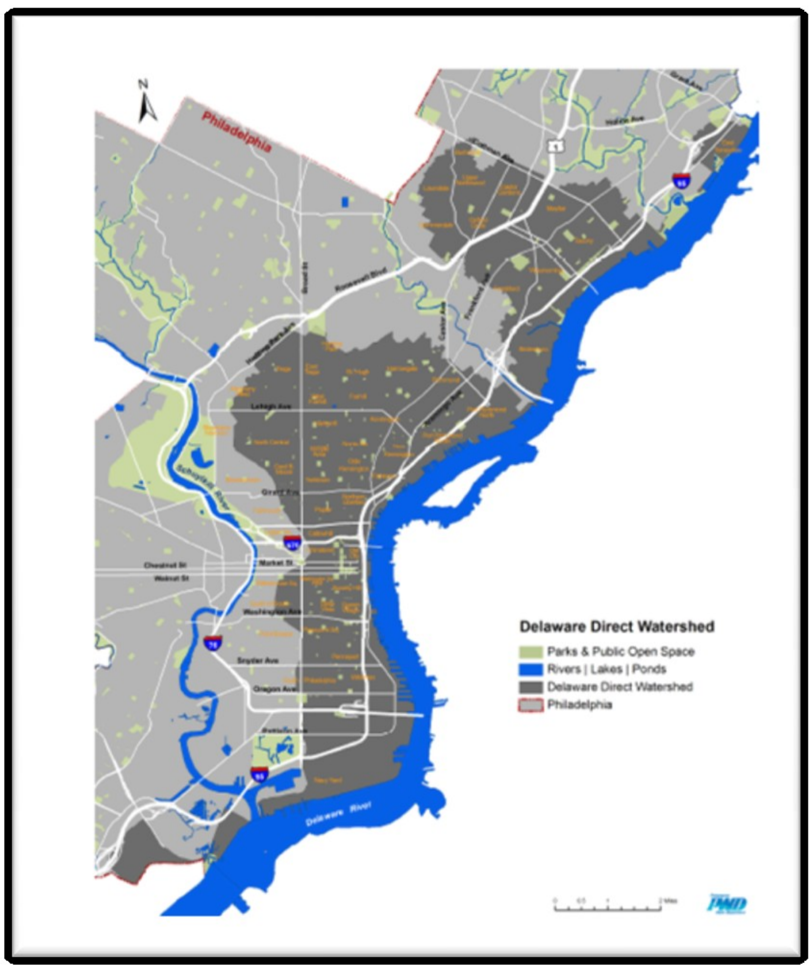

Figure 1-3 Delaware Direct Watershed Basin, Philadelphia region of Delaware River watershed (Source: PWD River Conservation Plan report, 2011).

The lower non-tidal Delaware reach located between NJ and the Delaware water gap (Covers Philadelphia) is intensively used for recreation, tourism, public water-supply and industrial purposes. One of the three Philadelphia water treatment facilities, the Baxter water treatment plant, is located on this River; and supplies over 190 million gallons of safe drinking water per day to citizens of Philadelphia and surrounding communities (PWD 2007).

Since the river and its tributaries constitute an interstate waterway (passing through New York, Pennsylvania, New Jersey and Delaware), a comprehensive regulatory body the “Delaware River Basin Commission” (DRBC) was formulated in 1961 marking a breakthrough in water resource management. This regional body is endowed with power of law to oversee a unified approach to managing the Delaware River without regard to political boundaries. As a framework for assessing the water quality, the DRBC divided the watershed in six interstate water quality management zones (WQM) and established designated uses for each segment of the river (Fig 4.) These designated uses are categories of ways in which the Delaware River is used by or provides value to people, such as support of aquatic life, recreation, public water supply and fish consumption (DRBWQA 2012). The boundaries of the WQM zones are shown in Fig. 4 and the third zone-“Zone 3” encompasses the Delaware Direct Watershed (Philadelphia region). According to the assigned designated uses, the water quality standards were developed separately for each zone by DRBC. Designated uses for zone-3 include aquatic life, public water supply, recreation-secondary only contact and fish consumption. The secondary contact activities as identified in DRBC Water Quality regulation include boating, fishing and other activities involving limited contact with surface waters incident to shoreline recreation. The parameters used by DRBC for assessing secondary contact recreation in the river are fecal coliform (maximum geometric average 770 per 100 milliliters) and enterococcus bacteria (maximum geometric average 88 per 100 milliliters) (DRBWQA 2012).

Figure 1-4 Delaware River Water Quality Management Zones (Source: 2012 Delaware River and Bay WQA report)

The Pennsylvania Department of Environmental Protection (PA-DEP) plays an extensive role in protection and restoration of this watershed by involvement in activities such as water conservation, aquatic life protection, discharge permitting, source and groundwater quality monitoring and enhancement, as well as encouragement and engagement of local watershed organizations. Out of multiple ongoing watershed planning efforts led by the Philadelphia Water Department, the Delaware Direct Watershed River Conservation Plan (RCP) is one of the most prominent efforts, run in collaboration with The Delaware Direct Watershed Partnership to bring together the non-profits, state/local stakeholders and community representatives to improve the overall environment of the watershed. The River Conservation Plan identifies significant natural, recreational and cultural resources; determine threats to river resources and recommend specific actions to conserve, and restore the project area. The Philadelphia chapter of the RCP is focused on direct drainages into the Delaware River within Philadelphia. The 2009 update report of the RCP indentified that almost 70% of the watershed area drains to the river via CSO system, leaving very few open water tributaries. Consequently, the RCP geared resources towards development of riverfront and neighborhood initiatives along the Delaware River. There are variety of workshops and programs seeking collaborations and partnerships across the local and state stakeholders for achieving wetland restorations, green streets with riverfront connections, improved parking lot designs and healthier neighborhoods.

There are several other ongoing planning initiatives in the Delaware Direct watershed for fostering collaborative efforts from public and private sectors to restore and manage the ecosystem and water quality of the watershed. Each plan is targeted to address the needs of the area it serves and makes recommendations for the enhancement of portions of the Delaware riverfront. Detail description of watershed management organizations and initiatives is provided at the PWD Research and Development database and can be assessed at http://www.phillywatersheds.org/your_watershed/delaware/delaware_RCP.

Watershed Population and Land use types: Overall, the population density of the Delaware Direct is high, around 14,764 persons per square mile (PWD 2011). However, the distribution of people within the watershed varies hugely showing great contrast in terms of population density. Large industrial and commercial facilities are clustered in the southern end of the watershed, while the central portion of the watershed is comprised of business districts. The northern and northeastern parts of the watershed are predominantly residential city neighborhood representing more even population distribution and relatively high population density. According to the 2010 U.S. census report almost 1,558,613 people live in the Philadelphia region of the watershed, the number was 1,191,579 in 2000 and 1,223, 066 in 1990, representing a drop of 2.6% from 1990 to 2000 followed by a spike of almost 30% in 2010 (PWD 2007; Kauffman 2011) .

The boundaries of the Delaware Direct Watershed are determined by drainage characteristics, not population patterns, and therefore it has irregular edges which are not confined by city blocks. According to the most recent- 2011 PWD watershed report, the most prevalent land use in the watershed is residential property (totaling nearly 40%). Most of this is comprised of row homes, residential multi-family and single-family housing. The next major component of land use is manufacturing and commercial property, taking roughly 23% of the land within the watershed. Transportation features, such as railways and roadways occupy approximately 10% of the watershed's land and another significant percent of land, around 8% is occupied by parking areas. One of the least represented features is the wooded area, reflecting the fast urbanized characteristic of this watershed (PWD 2011).

Another important feature of the urban watershed highlighting the major cause of stormwater runoffs and discharges from this watershed is the proportion of impervious surface cover, totaling 68% of the land area. Buildings, parking lots and travel ways make up the majority of impervious surface (almost 50%). Most riverfront land in the watershed is privately owned (totaling around 75%), providing challenges to the City in its effort for green and accessible riverfront development. Vacant land makes up 12.65 % percent of the total parcel area. A high concentration of vacant land in the watershed is located along the riverfront at former industrial and commercial sites (PWD 2011). The City conducts various workshops and programs to educate property owners, political leaders and the community about the economic, social and environmental benefits of extending the greenway along the length of the riverfront. The ultimate goal of the City is to develop an integrated system of trails, avoiding sensitive natural and cultural resources that would connect publicly owned lands within the watershed. As a recent development in regard to recreation and open space improvements in the watershed is the creation of Delaware River Waterfront Cooperation (DRWC). DRWC is a non-profit organization seeking to transform the central watershed region into a vibrant and easily accessible destination, thus significantly enhancing the probability of water recreation in the future. The other highly active non-profit organization is the Delaware River City Corporation (DRCC) which is also targeted to reconnect communities with the Delaware River by building greenways and parks along its shores. There are many other government agencies, non-profit organizations and concerned communities that have started to address the revitalization of the river and its communities.

1.1.1.3 Darby Cobbs creek watershedwatershed encompasses areas of Chester, Delaware, Montgomery, and Philadelphia counties, with parts of 31 municipalities and drains approximately 77 square miles to the mouth of Darby Creek at the Delaware Estuary. Most of the watershed is located in Delaware County. The watershed is further subdivided into the Cobbs

Creek, Darby Creek, and Tinicum subwatersheds for planning and management purposes (Figure 1.5). The only portion of the watershed that falls within the City limits is the section of Cobbs Creek subwatershed. The Cobbs Creek subwatershed is approximately 22 square miles (27% of the total watershed area) and it contains about 33 miles of streams (PWD and DCWP 2004).

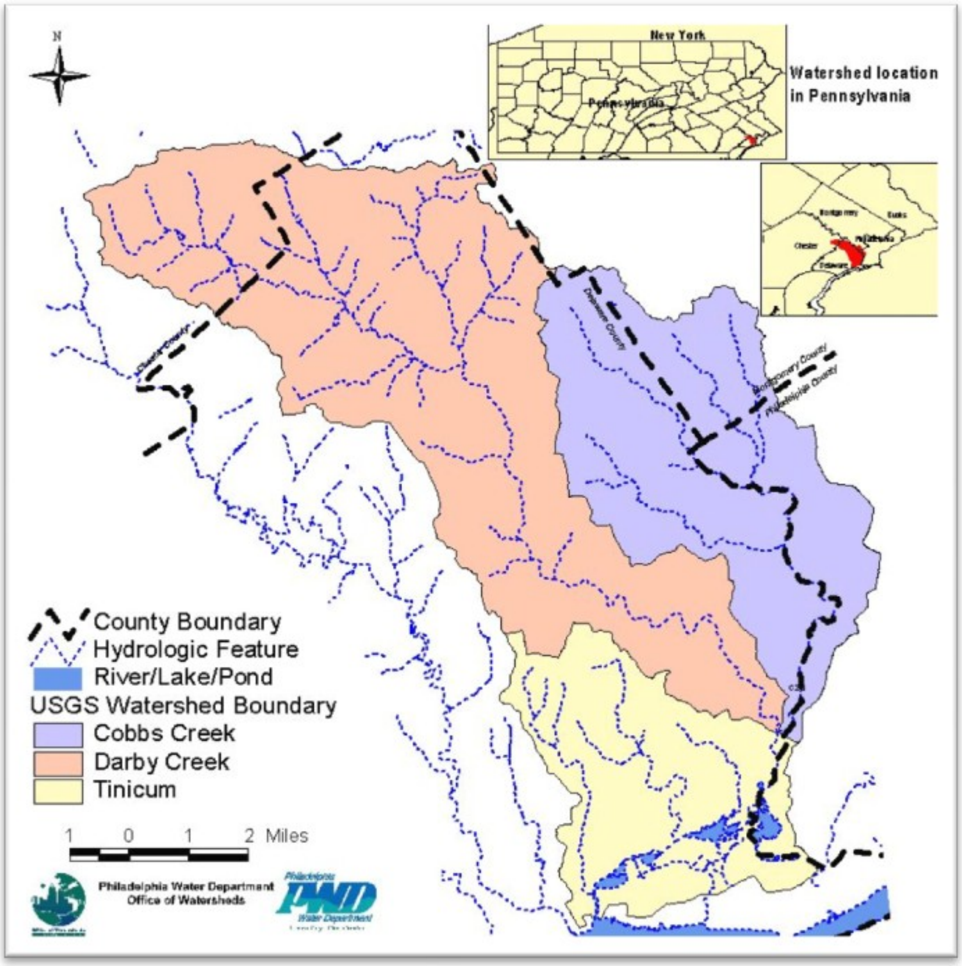

Figure 1-5: **Darby and Cobbs watershed with boundaries of subwatersheds (Source: PWD)**

The Cobbs Creek is located in West and Southwest Philadelphia; it runs the boundary of Phialdelphia and Delaware counties and is surrounded on both sides by the Cobbs Creek Park section of the Fairmount Park system. The upper portions and headwaters of Cobbs Creek, including East and West Branch Indian Creek, include portions of Philadelphia, Montgomery, and Delaware Counties. The lower portion of Cobbs Creek watershed, including the lower mainstem and Naylors Run, drain parts of Philadelphia and Delaware Counties. Cobbs Creek discharges to Darby Creek.

In 1999, the Philadelphia Water Department initiated the Darby-Cobbs Watershed Partnership in an effort to connect private and public entities as stewards of the watershed. Since then, the Partnership has directed various programs and led numerous projects for supporting subsequent planning and restoration activities within the watershed. The overall aim of the partnership is to assess the adverse impacts of land uses on surface and ground waters and thereby achieve higher levels of environmental improvement by sharing information and resources across all the active communities. An integrated watershed management plan (IWMP) was conducted by PWD in 2004 for the Cobbs portion of this watershed area; which outlined the approaches required to meet the challenges of watershed management in an urban setting. After extensive field study and data analysis the IWMP report identified 10 highest priority problems in the watershed, CSO impacts on water quality and stream channels being one of the major issues (PWD and DCWP 2004). Excess land development and urbanization are the major factors for the problems faced by the watershed. The water quality investigations conducted by PWD in 2005for characterizing the watershed condition, documented many violations of state water quality criteria, particularly in wet weather attributing it to discharges from CSOs. Damaged, improperly sized, or choked sanitary sewers and illicit connections exist across the watershed, leading to impairment of the waterway system in this watershed. The report identified that much of the watershed does not meet the state water quality standards for fecal coliform bacteria during dry weather and recommended investigation and abatement of dry weather sewage sources as one of the most important target for the watershed improvement plan (PWD 2005).

Another comprehensive watershed management program is the River Conservation Plan, completed by the Darby Creek Valley Association (DCVA) for the entire watershed drainage area in the year 2005 to broadly document the watershed's conditions at that time. DCVA's ultimate goal is to develop a 30 mile greenway system linking the entire watershed. In addition to these efforts, there are number of ongoing programs led by government and quasi-governmental agencies to improve the quality of the watershed, the description of those can be found at www.dcva.org website.

Watershed Population and Land use types: It was difficult to get the watershed population statistics for Philadelphia region of this watershed as the census districts do not follow the watershed boundaries. The average population density of the entire watershed is around 10 people per acre. Overall estimate for the entire watershed is approximately 500,000 persons, with as many as 166,143 residents in the Philadelphia portion of the watershed, as per 1990 census report. This number declined by almost 11,000 persons and reached to 155,447 in 2000 (DCVA 2002).

The decline in population was observed in many of the other smaller municipalities in the lower and middle portions of the watershed and reflected a variety of population dynamics in this watershed. The aging population with increases in deaths, reduction in average household size reflecting reduction in births, out-migration in general, out-migration of young people in particular, decline of employment opportunities, and other trends have been identified as the leading means for population loss in this watershed. Another important aspect of population is population density. The density in the City of Philadelphia is twice as great as that of any other municipality (24,138 persons/sq. mile), emphasizing the fact that it is one of the highly developed municipalities of this watershed (PWD and DCWP 2004). At the same time since the higher density developments generally overlook the environmental sensitive aspects of a watershed, it came with the high environmental cost.

The entire watershed is highly urbanized and almost completely developed. A major portion of the Darby-Cobbs watershed is covered by residential land use type, totaling up to 78.3% of the area. Wooded and recreational areas make approximately 3% of the area, resulting in 45% as impervious land area. The Delaware Valley Regional Planning Commission (DVRPC) have identified 24 land use categories in the Draby-Cobbs watershed and the RCP commission grouped that land use data into Upper, Middle and Lower sections of the watershed. Total 18 land use categories are identified for Philadelphia area of the watershed, with almost 70% of the land under residential use, 14% under commercial/community services, 3% vacant, 6% water and remaining 7% is distributed under parking, transportation, agriculture and wooded land use types (DCVA 2002). The percentage of imperviousness cover calculated by PWD for the city portion of the watershed is around 55.8%.

The land use pattern within the city changes drastically as one moves from the downstream portion to the upstream portion of the City. The lower portion comprises mainly commercial and industrial uses with major activities associated with airport-focused uses. Slightly upstream in the region of “West Philadelphia” the population density increases, with a larger proportion of the watershed occupied by older residential and commercial areas. Further upstream the watershed changes to a lower density neighborhood adjacent to City Line Avenue (PWD, PEC et al. 2009). As one of the major accomplishments of Fairmount Park Commission and dedicated volunteers from community, government and businesses an abandoned stable site in the watershed was converted into the Cobbs Creek Community Environmental Education Center (CCCEEC) in 2001. This historical building is one of the most scenic places in the City providing plethora of environmental benefits and opportunities for recreation and education to the community.

1.1.1.4 Tacony Frankford watershed The watershed covers portions of Montgomery county and Philadelphia County. The boundary of this watershed is defined by the land area that drains to the mouth of Tacony Creek at the Delaware Estuary, encompassing approximately 36 square miles in southeastern Pennsylvania with headwater region located in Montgomery County (PWD 2005) (Fig.6).

The creek is referred to as the Tookany Creek until it enters Philadelphia at Cheltenham Avenue, then it is renamed Tacony Creek until it joins the historical Wingohocking Creek by the Juniata Golf course. The Tacony Creek is referred to as Frankford Creek as it flows from Juniata Golf Course to the Delaware River just south of the Betsy Ross Bridge. The RCP for the Tacony-Frankford watershed is the major component of the greater effort to manage and restore the watershed and the section covered by PWD focuses on the watershed area that lies within the city of Philadelphia’s boundaries; which is approximately 12,230 acres(19 sq. miles) in size and represent 41.5% of the total watershed (PWD 2004).

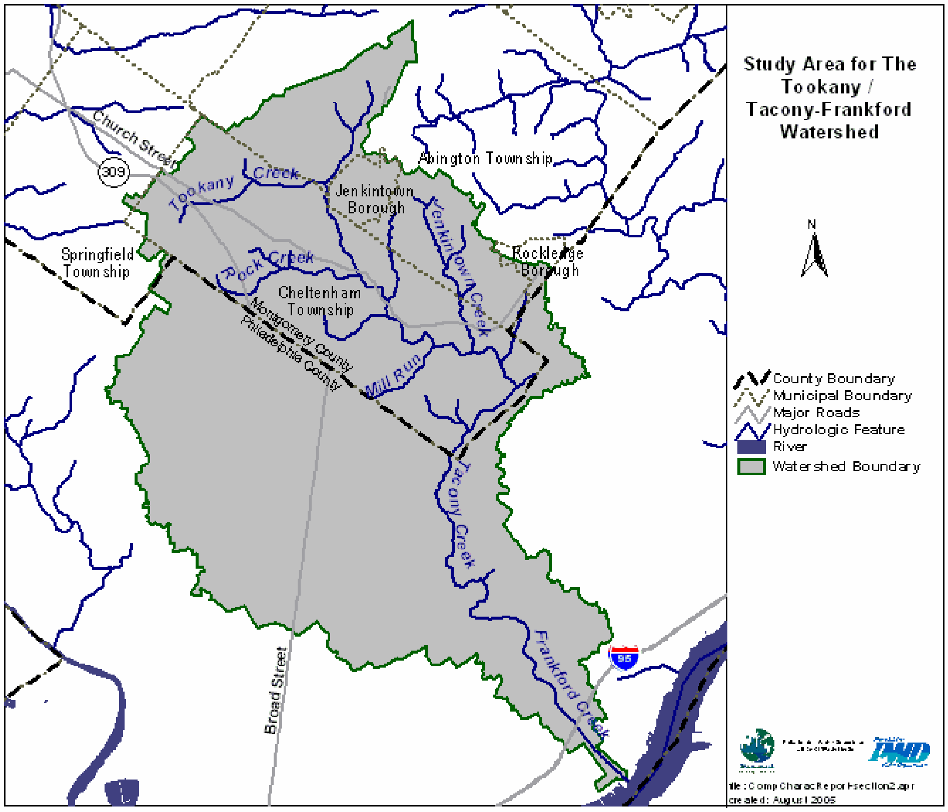

Figure 1-6 Tacony-Frankford watershed basin boundaries. (Source: PWD –RCP report)

The management of the watershed within the Philadelphia region is actively supervised by PWD. In order to connect diverse stakeholders as stewards of watershed, the PWD in 2000 launched the Tacony-Frankford Partnership (TFP) which in 2005 played a pivotal role in development of the Tacony-Frankford Integrated Watershed Management Plan (TF-IWMP). PA DEP assigns designated uses to water bodies in the state and performs ongoing assessments of the condition of the water bodies to determine whether the uses are met and to document any improvement or degradation. The Tookany/Tacony-Frankford Creek and its tributaries are designated warm water fisheries. All of the Tookany portion of the watershed plus tributaries, and the upper, non tidal portion of the Tacony Creek are classified as not attaining the designated use by PA DEP

(PWD 2005). Consequently, the state of the aquatic resource is required to be quantified periodically in order to get the stormwater permit for the City of Philadelphia.

PWD’s Office Of Watershed (OOW) is responsible for characterization and analysis of existing conditions in local watersheds to provide a basis for long-term watershed planning and management. An inventory of pollutant sources to the receiving water bodies was compiled by OOW in 2005 to analyze the relative contribution of each potential source (PWD 2005). Various types of sources were considered, including CSOs, industrial and process water discharges, septic tanks and atmospheric deposition. The number of septic tanks within the watershed is difficult to accurately quantify; according to 1990 census data there are approximately 1,075 septic tanks present in the watershed, 706 of which are located within the city of Philadelphia. Based on the federal and state NPDES database eight industrial and process water dischargers were identified in the watershed and by means of hydraulic simulation of sewer systems the portion of the flow that overflows to the receiving waters were predicted for CSOs. The analysis concluded that 90% of the fecal coliform introduced to the system is the result of CSOs; however this portion may change as the OOW analysis did not account for the contribution of Santitary Sewer Overflows (SSO) in the watershed. The other 6 major watershed issues as identified by the 2005 analysis are impaired wetlands, compromised water quality, lack of healthy riparian habitat and poor stream aesthetics due to trash, dumping and odors.

Watershed Population and Land use types: The upper reaches and headwaters of the Tookany/Tacony-Frankford Watershed are characterized primarily by a mix of multiple family and detached single-family residential areas, golf courses and parkland. The lower portions of the Tookany/Tacony-Frankford Watershed are primarily high-density residential areas in the City of Philadelphia, with commercial areas along highway corridors. Riparian lands within the City consist mainly of relatively undisturbed parkland. In 2005, the watershed assessment report indicated the Philadelphia region of the watershed has 53.6 percent impervious cover. The population of the entire drainage area, based on 2000 census data, is approximately 331,400 people. This yields an average population density of approximately 16 -17 persons/acre (PWD 2005). The population density by county indicates the Philadelphia County is more heavily populated than Montgomery County, as number estimated for Montgomery County is 7 people/acre versus 24 people/acre for Philadelphia County. From 1990 to 2000 the watershed observed 3% decline in the population, which is largely due to demolition of homes in the Logan neighborhood (PWD 2004; PWD 2005).

As mentioned above, the drainage area of this watershed receives contributions of significant amounts from both point and non-point source discharges, in addition to CSOs that impact the water quality. According to the USGS data for the study area, the breakdown by sewer type is as follows: combined sewer areas make up 9,800 acres, or 47% of the drainage area; separate sewers, including areas outside of the City of Philadelphia, account for 9,200 acres or 44% of the drainage area; and non-contributing sewers make up 1,900 acres or 9% of the drainage area (PWD 2005).

The Tacony-Frankford watershed within the City does not have any agricultural land area. Wooded, recreational and park area comprise approximately 10% (1,210 acres) of the land, residential units cover almost 60% area, transportation 6% and commercial/manufacturing services spreads in 16% of the watershed area. There are approximately 313 acres of vacant land under the City side of the watershed, primarily located in Hunting Park, Frankford, Richmond and Bridesburg (PWD 2004). Philadelphia City Planning Commission (PCPC) is actively conducting outreach and community efforts for revitalization of the City. Since 2002, PCPC have initiated 4 major redevelopment projects for vacant and brownfield areas in this watershed, including Logan, Frankford Creeek area, Ogontz neighborhood area and Gearter Germantown Housing development programs. These economic redevelopments of the watershed's neighborhood would lead to improved environmental health of the watershed and encourage watershed stewardship at a larger extant.

1.1.2. Combine Sewer Overflows and Storm water management policies

Philadelphia is served by two types of sewer systems. About 60% of the City's sewered area is served by combined sewers, typically the older sections of the city. The other 40% of the sewered area is served by separate sewers (Fig. 1.7). The combined sewers collect both stormwater runoff and the wastewater from homes, businesses and industries. Under dry weather conditions and during smaller storms the wastewater/stormwater mixture is carried to the treatment plant via the combine sewers for treatment prior to discharge of the treated effluent into the waterbodies. However the sewers get inundated with large quantities of stormwater mixed with wastewater during the large storms at a rate beyond the capacity of the treatment plant or interceptor sewer. The volume in excess of the treatment plant capacity is released to a nearby stream/river without treatment and is referred to as Combine Sewer Overflow (CSO). There are 164 CSO's in Philadelphia discharging to 5 different waterbodies within the city. In addition to these outfalls there are approximately 455 separate stormwater outfalls discharging only the surface runoffs from the street inlets, building downspouts, and other storm sewer lines to the nearby receiving streams during the heavy storm periods (PWD 2012).

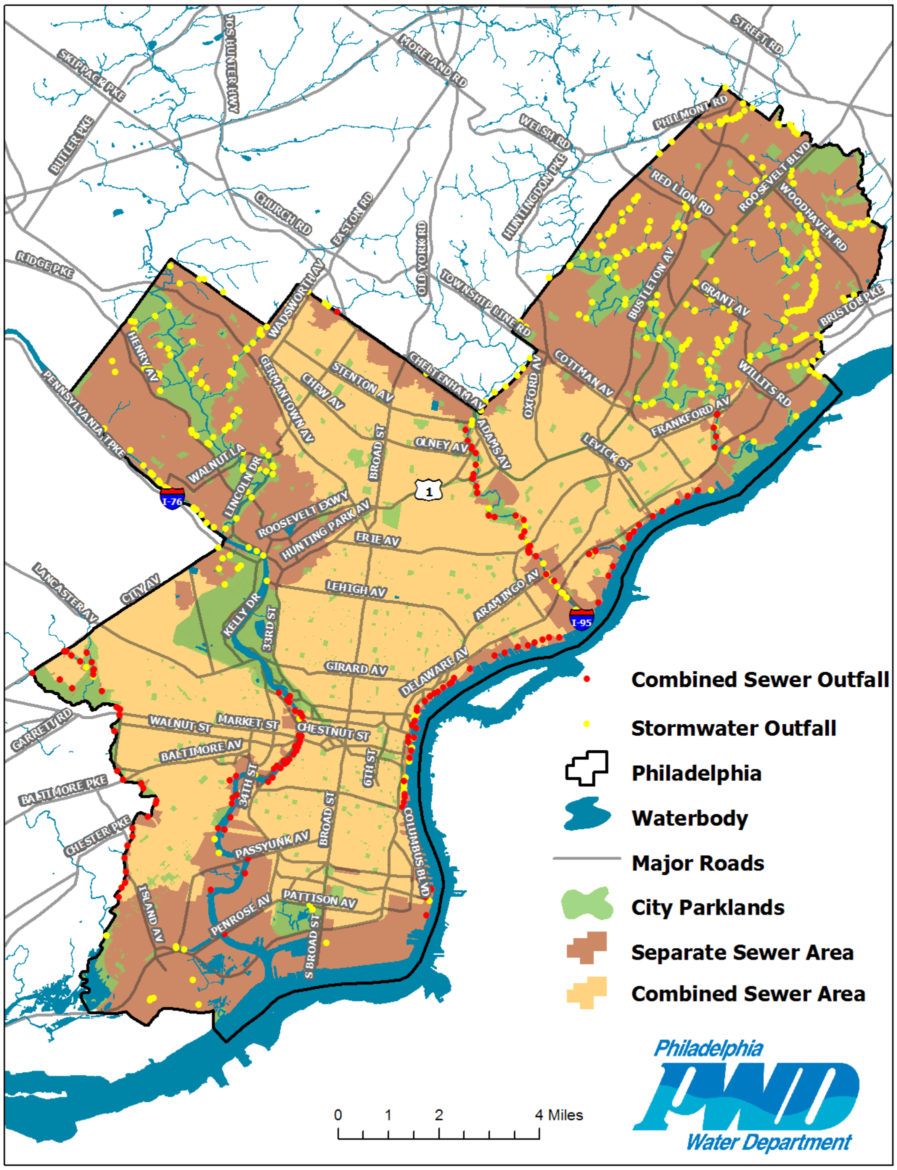

Figure 1-7: **Distribution of CSO served area across the City (Source: PWD)**

During heavy storms, the overflows from combine sewers may cause the receiving water bodies to exceed the water quality standards impairing the use and enjoyment of that water system.

These CSO's are considered as the point sources of pollution and are subjected to National Pollutant Discharge Elimination System (NPDES) permit requirements including both technology-based and water-quality based requirements of CWA.

According to the summary report from PWD for year 2006-2009, the largest annual CSO discharge volume was approximately 1.7 BG, with total volume of discharge reaching to 14 BG (Crockett 2010). On an average there are approximately 65 rainfall events per year in Philadelphia obtained from the 1902 to 2005 precipitation records at PWD/USGS. The occurrence of the overflow from a combine sewer system depends on various conditions, most importantly on the size of the conduit and the drainage area, amount of precipitation, land use type and topography. Thus the quantity and frequency of discharge from each CSO is unique, but the Office Of Watersheds (OOW) at PWD developed hydraulic and hydrologic models to simulate the runoff patterns and provide annual estimates of such discharges. Table 1.3 provides the list of permitted CSO outfalls and number of major stormwater outfalls on each waterway within the service area of PWD, as identified in the City's NPDES permits (PWD and NPDES 2010).

Table 1-3 Combined Sewer Overflow and storm water point source discharge counts.

Water body	No. of CSO outfalls	Number of storm water outfalls
Schuylkill River	33	42
Delaware River	61	20
Cobbs Creek	34	3
Tacony Creek	31	35
Pennypack Creek	5	130

(Source: PWD-Combined sewer management program annual report, 2010)

A national framework for regulations of CSOs under NPDES is provided by EPA through its Combine Sewer Overflow Control Policy, published in 1993 (U.S. EPA 1993). The Policy guides municipalities in meeting the pollution control goals of the CWA in as flexible and cost-effective a manner as possible. As part of the program, communities serviced by combined sewer systems are required to develop long-term CSO control plans (LTCPs) that will result in full compliance with the CWA in the long term, including attainment of water quality standards. PWD completed its LTCP in 1997 and is currently implementing its provisions.

The Philadelphia Water Department (PWD) is involved in myriad watershed planning efforts with an ultimate goal to achieve source water protection and to develop sustainable water and storm water infrastructure. As discussed earlier the CSO discharges have been reported as the dominant source of pollutants in the River Conservation Plan reports of all the seven watersheds. Thus, PWD conducted a comprehensive study to evaluate stormwater management alternatives and finally selected the green stormwater infrastructure based approach as the best approach to address the CSO concern of the City. The "Green City, Clean Waters" approach, proposed by PWD in 2009, is a 25 year program to achieve environmental, economical and social restoration of the City within the most efficient timeframe (PWD 2009). As oppose to traditional approaches such as rebuilding the new sanitary system/treatment plant with increased capacity or providing separate sewer system in the entire city, this approach targets on physical alterations to properties to enhance water infiltration into the ground. In an attempt to encourage better on-site stormwater management, the City has also changed the storm water billing design from a meter-based system to impervious area based system. Under this provision, the private landowners who reduce impervious areas can receive credits up to nearly 100% on their storm water bill (PWD 2009).

The revolutionary work done by PWD in revitalizing the city since the inception of this plan has helped the city rank first in the Natural Resources Defense Council's 2011 November report on green infrastructure (Garrison, Hobbs et al. 2011). The city is testing a variety of designs and technologies such as porous pavement, rain gardens, green roofs, storm water tree trenches and vegetated bump-out curb extensions to transform 9500 acres of impervious cover - about half of the total drainage area - into green acreage, and to restore 20 miles of urban stream corridor.

1.1.3. Recreational water quality regulations

In 2000, Congress passed the Beaches Environmental Assessment and Coastal Health (BEACH) Act amending some sections of CWA to address pathogens and pathogen indicator in coastal recreational waters. This act also required the EPA to issue new or revised recreational water quality criteria by October 2005 (U.S. EPA 2000). However, the agency failed to meet mandated deadlines to update its water quality standards based on new research and to develop new tests that provide same-day results. It faced a federal lawsuit in 2006 by the Natural Resources Defense Council (NRDC) as a record number of beach closings and warnings due to unsafe beach water were reported during that time (Negin and Heyd 2006). EPA reached a settlement with NRDC in 2008 and agreed to develop new pollution testing methods and public health standards that would protect against a broad range of swimming related illnesses, including skin rashes, pinkeye, respiratory illnesses, ear infections, as well as more serious illnesses such as hepatitis and meningitis (Dorfman and Rosselot 2011). Current standards are based only on gastrointestinal ailments commonly known as the stomach flu (U.S. EPA 1986) and the testing methods take 24-48 hours to produce results, so the information on whether the beach is safe for swimming gets to the people at least a day after the exposure. The agency released the 2012

Recreational Water Quality Criteria draft document (RWQC) in December 2011 which was under external scientific review till February 2012. Current date for new criteria is October 15, 2012, per Consent Decree (U.S. EPA 2011).

1.1.3.1. **History of Water Quality Criteria Development:** In 1948, U.S. Public Health Service (USPHS) conducted epidemiological studies for health hazard in swimmers and found elevated GI cases when Total Coliforms (TC) exceeded 2300 CFUs/100ml (Stevenson 1953). Based on this finding, the National Technical Advisor Committee (NTAC) proposed the first water quality criteria in 1968 (NTAC 1968). They recommended fecal coliform (FC) as indicator organism and translated the US PHS's TC level to 400 FC/100ml based on a ratio of TC to FC derived from a water quality study done in Ohio. For better protective measure they further halved the FC number to 200 FC per100ml and proposed it as the allowable 30-day geometric mean value with no more than 10% samples in 30-days period showing concentrations greater than 400 cfu/100ml. This criterion was believed to represent zero risk at that time. The NTAC criteria for recreational waters were recommended again by EPA in 1976 (U.S. EPA 1976).

In the late 1970s and early 1980s, EPA initiated a series of epidemiological studies to evaluate several additional organisms as possible indicators of fecal contamination. They concluded that associations between GI illness and *E. coli* and enteroccoci are better indicators than fecal coliforms and so recommended them as new indicators for the water quality criteria in 1986 (U.S. EPA 1986). The 1986 criteria values were developed to be "as protective as" the 200 CFU/100ml FC criteria by maintaining the same water quality criteria, converting FC to enterococci using ratios of observed water quality data (GM: Geometric Mean and SSM: Single

Sample Maxima) from epidemiological studies. This approach resulted in separate illness rates for marine and fresh waters because the marine and fresh water studies reported different GMs for the FIB associated with the level of water quality corresponding to EPA's FC criteria recommendations. The final recommendations of the 1986 Ambient Water Quality Criteria for Bacteria for Highly Credible Gastroenteritis Illnesses (HCGI) are given below in table 1.4.

Table 1-4 1986 Ambient Water Quality Criteria recommendations

Waterbody-FIB	Illness per 1000 swimmers	30-day Geometric Mean	Single Sample Maximum (Confidence Limit)		
			Upper 75%	Upper 90%	Upper 95%
Freshwater- EC	8	33	235	410	576
\Freshwater- ENT	8	126	61	107	151
Marine water- ENT	19	35	104	275	500

1.1.3.2. **2012 Recreational Water Quality Criteria Recommendation**: In order to support the development of new or revised recreational water quality criteria by October 2012, EPA conducted epidemiological studies at marine and fresh waters in both temperate and tropical regions of United States from 2003 to 2009 (U.S. EPA 2011). These studies were prospective cohort studies and enrolled 54,250 participants encompassing 9 locations; as a group they are referred to as the "National Epidemiological and Environmental Assessment of Recreation", NEEAR studies. These studies employed both the culture based (MF: Membrane Filtration) and DNA amplification based (qPCR: quantitative Polymerase Chain Reaction) methods in determining Fecal Indicator Bacteria (FIB) densities in study waters. These studies also evaluated other adverse health endpoints that could have been associated with pathogens found in fecal matter. These included: upper respiratory illness, rash, eye ailments, earache and infected cuts. But based on their analysis they determined that the criteria protecting the public from GI illness will prevent most types of recreational waterborne illnesses and continued with recommendations for GI related illness.

The ideal approach to set up new criteria would be to compare the existing 1986 Bacteria Criteria recommendations to NEEAR studies; however EPA was unsuccessful in its search for the raw individual data points needed to conduct this comparison. Some of the data of 1986 epidemiological studies were stored on magnetic tapes that had degraded with no source within EPA or outside EPA to obtain the data (Noss and Ravenscroft 2009).

The major outcomes of the 2012 RWQC draft are as follows:

- Development and validation of a qPCR method as a rapid analytical technique for testing enterococci in recreational waters. Though the study waters demonstrated negligible interference, EPA recommends a site specific assessment of qPCR performance with respect to sample interference prior to adopting it into State WQS for implementation in beach monitoring programs due to their limited information for inland and tropical marine waters.
- EPA retained the culture method as the testing method for *E. coli* and enterococci in freshwaters and enterococci in marine waters, but replaced the single sample maximum with Statistical Threshold Value (STV).
- EPA' recommendations no longer include a recommendation to calculate the GM criterion over a period of 30 days. The duration should range between 30 to 90 days.
- For States interested in adopting a value for enterococci using EPA's *Enterococcus* qPCR method (Method A) into their WQS, EPA recommends a *Geometric Mean* criterion of

475 CCE per 100 mL and an *Statistical Threshold Value* criterion of 1,000 CCE per 100 mL in freshwaters and marine waters based on its epidemiological study data.

- EPA indicated that freshwater and marine data shows that illness rates are similar in both water bodies and recommended acceptable illness level of 6 to 8 cases of Highly Credible Gastroenteritis Illness (HCGI) per 1000 recreators in both waters.

The summary of 2012 RWQC is shown in table 1.5 below.

Table 1-5 Summary of 2012 RWQC recommendations

Indicators	Clean water act purposes			Beach monitoring purposes		
	Illnesses per 1000 swimmers	Geometric Mean (GM)	Statistical Threshold Value (STV)	Culture method (STV)	qPCR method A (CCE per 100 mL)	
					GM	STV
Freshwater- EC	8	126	235	235	-	-
Freshwater- ENT	8	33	61	61	475	1000
Marinewater- ENT	8	35	104	104	475	1000

The EPA suggests that under the current state of knowledge the qPCR methods should be applied to high risk beaches only and it will provide the implementation recommendations in the forthcoming technical support materials.

1.1.3.3. **Current State of recreational water quality regulation in Pennsylvania** In general about one-third of all states in USA have adopted either *E. coli* or enterococci as fecal indicator organism to represent the water quality of fresh and marine waters (U.S. EPA 1999). Other states, including Pennsylvania continue to use total coliforms as the fecal indicator to monitor their recreational waterways.

Pennsylvania's water quality standard for fecal coliform bacteria in recreational waters is as follows: during the swimming season (May 1 through September 30), the maximum fecal coliform concentration shall be a geometric mean of 200 CFU per 100 mL based on five consecutive samples each sample collected on different days; for the remainder of the year, the maximum fecal coliform level shall be a geometric mean of 2000 CFU per 100 mL based on five consecutive samples collected on different days (PWD 2005).

The statewide water-uses identified under Chapter 93 of the Pennsylvania Code (Pennsylvania Code 1971) characterizes surface water recreation in three categories – boating, fishing and water contact sports (§ 93.4). Specific water quality criterion (§ 93.7) is recommended only for water contact sports, which refers to swimming and related activities. Though the criterion is protective of water quality, it does not reflect the specific risk of illness posed by recreational use of a particular water body. A different approach to judge if the waters are safe for recreation would translate the bacterial water quality into the associated health risks posed by humans upon exposure to such waters. This approach is based on epidemiological studies and results in a dose-response relationship. Using such mathematical models, it can be correctly identified if the city water will pose any health risk to its users, and if so, to what extant will that be associated with wet weather CSO discharges.

The city has faced number of water quality challenges in the last few decades, largely associated with CSO discharges leading to violation of state and federal regulations. Consequently, the Philadelphia Water Department is evaluating different techniques to reduce these flows as well as to get an estimate of health hazard from recreational exposure to these waters under wet weather discharge conditions. As a component of this effort, the department initiated this study with an aim to identify usage patterns along the water bodies within the city and to examine the relative contribution of each observed activity in causing gastroenteritis illnesses among the users.

1.1.4. Recreational exposure and diseases

Swimming is the fourth most popular recreational activity in the United States as per the recent outdoor recreation participation report of the 2009 U.S. census. Fishing at fresh waters and boating are also ranked in the top 10 common outdoor recreational activities for Americans (U.S. Census Bureau 2012). Within Pennsylvania there are ample opportunities for outdoor water recreation as it has more than 86,000 miles of streams and rivers and 160,000 acres of lakes. Other than swimming, the common uses of coastal and fresh water environments in Pennsylvania are boating, fishing, canoeing, white-water rafting, kayaking, and jet skiing (PA DEP 2006).

All the different modes of water recreation presents potential for recreational water illnesses (RWI) as the exposure to pathogens can be through swallowing, breathing in mists or aerosols or having physical contact with contaminated waters in lakes, rivers or oceans. There is a wide variety of infections that fall under the category of RWIs; including gastrointestinal, skin, ear, respiratory, eye, neurologic, and wound infection (CDC 2010). The most frequent adverse health outcome associated with exposure to fecal contaminated recreational water is gastroenteritis illness. These illnesses are caused by pathogens such as *Cryptosporidium, Giardia*, *Shigella*, norovirus, and *E. coli* O157:H7 and affect a person's stomach and intestines causing diarrhea, nausea, or vomiting.

Since 1978, the Centers for Disease Control and Prevention (CDC), EPA and the Council of State and Territorial Epidemiologists have collaborated on the Waterborne Disease and Outbreak Surveillance System (WBDOSS) for collecting and reporting data on waterborne disease outbreaks associated with recreational water (Craun, Craun et al. 2006). This surveillance system is the single most complete source of information concerning the scope and health effects of waterborne disease outbreaks in the United States. It is published every two years and the most recent summary was published in 2011 summarizing the outbreaks that occurred during 2007 to 2008 (Hlavsa, Roberts et al. 2011). It reported around 134 recreational water outbreaks resulting in at least 13,966 cases, the largest number of outbreaks ever reported in a 2-year period. Of the 134 outbreaks, 81 (60.4%) were outbreaks of acute gastrointestinal illness (AGI); 24 (17.9%) were outbreaks of dermatologic illnesses, conditions, or symptoms; 17 (12.7%) were outbreaks of acute respiratory illness (ARI); one (0.7%) of ear-related illnesses, conditions, or symptoms; and one (0.7%) of other illness. The remaining 10 (7.5%) outbreaks were of combined illness types, nine of which included ARI. Outbreaks of AGI accounted for 12,477 (89.3%) of the total outbreak-related cases; 64 (79.0%) of the 81 AGI outbreaks started during June, July, or August and resulted in 12,477 (89.3%) cases.

The exposure route implicated for each outbreak listed in the 2011 surveillance report was ingestion for 81 outbreaks (60.4%), contact for 25 (18.7%), inhalation for 18 (13.4%), combined routes for six (4.5%), and unknown for four (3.0%). Other illnesses related to recreation exposure to waters include otitis, conjunctivitis (Dewailly, Poirier et al. 1986; Yau, Wade et al. 2009), swimmer's itch (Verbrugge, Rainey et al. 2004), hepatitis A (Mahoney, Farley et al. 1992), Leptospirosis (Narita, Fujitani et al. 2005), meningoencephalitis (Cogo, Scagli et al. 2004), and Pontiac fever (Modi, Gardner et al. 2008).

However the 2012 recommendation of EPA ascertaining that the waters protective of AGI are protective of all other recreational water illnesses, structured the present work to address only GI cases in evaluating the health risk status of Philadelphia waters. In addition to this, the continuous acceptance of *E. coli* and enterococci as preferable fecal indicator organisms for fresh water bodies suggested their use as indicator organisms in this study. The purpose of using different indicator organism than the currently used by the Pennsylvania DEP (fecal coliforms) is to give the public health officials and agencies responsible for maintaining use in water used for recreation a new perspective for evaluating the environmental impacts of CSO discharges that may lead to potential health risk to the users. Using *E. coli* and enterococci as indicators would allow the application of their established dose-response models into the risk assessment calculations and thus would generate quantitative health risk estimates. This information would be beneficial to the city officials and the decision makers when they are investigating different measures to mitigate CSO discharges.

1.2 Dissertation Objectives and framework

1.2.1 Dissertation Objectives

The overall objective of this dissertation is to identify if the recreational water of Philadelphia pose an excessive health risk to its users, and if so, what is the extent of the risk during the periods of wet weather discharges from Combine Sewer Overflows (CSO). Sites located close to an existing CSO and having significantly high usage potential were monitored during the peak recreational season (May through September) from 2008 to 2010. A map detailing location of the sites chosen for this study is shown in Figure 1-8. Five sites (PH: Pleasant Hill park, PP: Pennypack park, FA: Frankford arsenal, PT: Penn Treaty park and PL: Penns Landing) are located in the Delaware Direct watershed, 2 sites (FD: Fairmount Dam and BG: Bartram's Garden) are in the Schuylkill river watershed, 3 sites (AD: Adams avenue dam, BH: Bingham street and T14) are in the Tacony-Frankford watershed and one site (COB: Cobbs environmental education center) located in Cobbs creek watershed.

Figure 1-8: **Recreational use monitoring locations during 2008 – 2010 recreational season.**

The study focuses on three major objectives in order to quantify the recreational health risk to Philadelphia users. The first objective is to understand the current recreational use pattern at Philadelphia waters. For this purpose, a novel surveillance method is tested which is less labor intensive as compared to traditional methods (in-person counts) and allows to gather a large number of continuous observations (irrespective of the weather or time of day) without causing any disturbance to the users. The goal under this exercise is to collect comprehensive exposure data which can be analyzed to generate probability distributions for the durations of all observed recreational activities.

The second objective is to address the concern about the increased health risk under combined sewer overflow conditions. For this purpose, the approach of quantitative microbial risk assessment is used. The collected recreational use information along with the water quality data at each site is synthesized to predict the number of expected GI illnesses at each study location, under dry and wet weather conditions. This provided an insight into the weather influence on the health risk of the recreational users, which is critical to know when the city is making massive investments to mitigate CSO discharges. The analysis is further expanded to predict the risk of illness per 1000 users per day, to identify if Philadelphia water bodies are in compliance with 1986 criteria for risk level lower than 0.8% of swimmers (U.S. EPA 2004). Water quality data are analyzed for both the fecal indicator bacteria's (FIB) – *Escherichia coli* and *Enterococci* spp. as both of them are the recommended indicator organisms for freshwater studies and have published dose-response models to be used in risk assessment.

The third objective focused on analyzing if the local risk estimates at the study locations are representative of the overall public health status from exposure to the entire water body. This was achieved by first scaling up the usage profile by developing regression models for camera monitored sites and then by estimating the total risk to the users across the entire stretch of the water system. Given the scale of the current research and limited availability of the resources, this objective is targeted for the Schuylkill and the Delaware River only. Expected recreational use preferences in the unmonitored stretch of the river mile are obtained from the watershed officials through in-person interviews and online survey. Health risks associated with the level and types of recreation most commonly occurring on the rivers are estimated using the projected use patterns. The overall goal was to identify the critical zones across the rivers that present significant health risk as compared to the rest of the river length. This information is crucial to the allocation of existing resources and is also useful in planning and budgeting the development of new facilities.

1.2.2 Scope of the work

The following steps broadly define the scope of the present work:

- Develop a recreational use data collection strategy using time lapse construction cameras.
- Develop probability distribution functions characterizing the activity duration data and the user frequency data for each observed activity.
- Analyze the influence of weather and time of the day on the usage preferences of the users.
- Identify the type of recreational activity contributing maximum to the total health risk observed to the users at each study location.
- Compare the number of GI illnesses under dry and wet weather conditions, using enteroccoci and *E. coli* as fecal indicator bacteria to analyze the influence of CSO discharges on public health.
- Identify the locations exceeding the de facto recreational health risk standard of EPA (more than 8 excess illnesses/1000 users/day).
- Analyze the sources of uncertainty and variability in the risk estimates.

- Evaluate the total recreational use demand and the total health risk status across the entire river mile within the city, for the Delaware and the Schuylkill River.

2 Chapter 2: Recreational use Assessment of Water-based Activities, using Time-Lapse Construction Cameras

2.1 Introduction

The decision to treat (disinfect) wastewater and storm water overflows is a major controversy in the U.S. and will probably cause large cities massive investment in "gray" infrastructure to mitigate wet weather discharges. A rational decision would use risk-benefit or cost-benefit approaches. To assess risk, one must know how the waters are being used. While in many cases where the dominant use is swimming, the quantification of exposure is more straightforward, e.g. by using aerial photos, turnstile counts or parking fees etc. In a context such as the one described in this paper where non-swimming recreational activities are the principal mode of exposure to surface water bodies, the issue is both determining how many people are using a water and for what purpose (swimming, fishing, etc.). From this information, the exposures can be computed and thereby the ultimate risk.

Urban water-based recreational experiences can vary significantly both temporally and spatially across a watershed. Factors such as geographic heterogeneity, accessibility and development, population composition and density all contribute to the diversity of recreational use, which affects the duration and type of actual water exposure, ingestion volume, and subsequent associated risk of developing waterborne pathogenic illness. Studies have shown that despite the increasing demand of non-swimming water recreation (Cordell 2004), the information on associated health risks (including multiple pathways) is minimal (Dorevitch, Panthi et al. 2011). Variability in exposure produces significantly different evaluations of risk assessment. So improvements in methods for accurately capturing recreational use exposure are vital to producing more accurate epidemiological studies and microbial risk assessments.

However, currently no standard protocols exist for the collection of data on non-swimming recreational use frequency and activity patterns. Previous studies on microbial risk assessment have attempted using a variety of approaches including field surveys, visitor questionnaires, traffic counters, voluntary registration and remote sensing. These traditional methods are labor intensive and are subject to several potential biases. Surveys conducted in the field, either by questionnaire or by visual observation may interfere with the users' actual activity pattern. Significantly, in most cases investigators do not get a continuous observation and have to characterize the usage based on sporadic data collected during the survey duration.

Multiple studies have reported usage frequency for swimmers and non-swimmers based on records of lifeguards stands (Turbow, Osgood et al. 2003; Given, Pendleton et al. 2006), or conducted randomized trials for evaluating exposure differences between bathers and non-bathers (Kay, Fleisher et al. 1994). At least one study used aerial photographs in conjunction with interviews of state officials (Soller, Olivieri et al. 2003). Moreover, most of the recreational use studies focus on generating annual usage statistics and lack addressing periodic variations on a seasonal, weekly or daily basis (Sidman, Swett et al. September 2005).

Primarily, most studies have defined risk-associated recreational activities for only two broad categories: swimming or non-swimming, and focused on beaches and marine environments rather than freshwater bodies (Prüss 1998; Wade, Pai et al. 2003). However, there are a few exceptions; *Cryptosporidium* risk was assessed for urban anglers through questionnaires, hand washing and fish sampling (Roberts, Silbergeld et al. 2007) and risk of gastrointestinal (GI) illness among surfers was researched by Stone and other authors (Stone, Harding et al. 2008). Limited contact recreational activities, such as boating, fishing, kayaking, wading, canoeing, jet skiing etc. have not been fully researched, because it is assumed that many of these activities have lower rates of exposure to water and thus to gastrointestinal pathogens compared to swimming. The only study found to address the human health risk associated with secondary contact water uses was published recently for the Chicago Area Waterway System (CAWS) (Dorevitch 2011). This comprehensive multi-stage research program entitled, "Chicago Health, Environmental Exposure, and Recreation Study (CHEERS)" is the first U.S. Epidemiology study to address the secondary contact exposure risks, and provide a spectrum of information necessary to conduct detailed non-swimming recreational exposure assessment (Geosyntec 2008; Rijal, Petropoulou et al. 2009; Rijal, Tolson et al. 2011). In addition to the detailed epidemiology study, this program also provided the thorough review of the USEPA 's 1986 Ambient Water Quality Criteria for Bacteria as applied to secondary contact recreation (Kollias 2006). In particular, the data for recreational use classifications for the types and frequency of recreational exposures expected in the CAWS was derived from the Use Attainability Analysis (UAA)survey, conducted by the Illinois Environmental Protection Agency (IEPA) from 2002 through 2007 (Kollias, Granato et al. 2008).

In contrast to water-use studies, wildlife and ecology researchers have been using time-lapse cameras for more than 50 years to record behavior patterns in animals, often at remote and/or virtually inaccessible locations. In 1877, the first documented use of remote photography involved a galloping horse triggering a line-up of cameras through tripwires illustrating that all feet were lifted above-ground simultaneously (Guggisberg 1977). For example, the use of time lapse photography to study raptor nesting behavior can be dated back to 1970s (Booms and Fuller 2003). The prevalence of this technology was demonstrated by Cutler and other authors (Cutler and Don 1999), who evaluated the efficacy of utilizing remote photography through a meta-analysis of 107 papers that had used it to study vertebrates in the field.

Other professions, such as the medical field, have adopted photographic monitoring for detecting activity pattern disorders (Jansen, Rebel et al. 2008). Occupational hygiene researches used the technology to assess activities and real-time exposure to chemicals or substances (Gressel, Heitbrink et al. 1987; Andersson, Niemelä et al. 1993; Hakkola 2000; Rosen 2005; Kaur, Clark et al. 2006).

The lack of a standard method for the collection of precise recreational use data, coupled with the temporal and spatial variation in recreational use patterns at multiple locations within the metro area will be addressed in this research. In particular, emphasis is on testing the capability of novel "time lapse photography" technology for gathering water exposure data. The main concern in using this technology is whether it is as reliable as in- person water-recreational use data collection. Typically, a survey methodology would be evaluated as to how accurately it can develop probability distributions for the duration of all observed non-swimming recreational activities. There were two primary objectives of this study. The first was to assess the efficiency of remote surveillance to collect recreational use data, and the second objective was to model exposure distributions to provide input data for computing the associated risk of illness. For this purpose, cameras were used (in conjunction with in-person surveys) to accurately identify the number of recreational users and to characterize their activity patterns at each location for weekends and weekdays, and during wet and dry weather. During wet weather events, storm water runoff from agricultural, industrial, and metropolitan lands and discharges from combined sewer overflows (CSO) carry high loads of untreated sewage, organic, and inorganic wastes into receiving waters, affecting water quality.

The analysis presented here is, to our knowledge, the first exposure assessment for non-swimming recreational water experiences that uses camera captured images to quantify human exposure to surface waters. Another similar study that used camera images for exposure assessment (Wang, Solo-Gabriele et al. 2010), counted human bathing events and animal visitations from 2005 to 2008, at a beach site for assessing *Enterococci* loads. The principal limitation of the study was related to the large picture intervals (15 minutes) and lack of analysis for the effect weather conditions had on variation in population and activity. Our approach to characterizing several types of exposure activities and duration using time lapse cameras for evaluating human health outcomes is novel. The exposure estimates will directly inform decision-makers in the metro area as to future infrastructure development.

2.2 Materials and methods

2.2.1 Recreational water sites

Eight sites located close to an existing combined sewer overflow (CSO) pipes within the city of Philadelphia, with a significantly high usage potential were monitored using time lapse cameras during the peak recreational season (May through September) in 2008 to 2010. The sites were selected based on pilot surveys and interviews with the local governing authorities who were familiar with each area's patronage. The volume of discharge from CSO outfalls was not taken into account when generating estimates of exposure. Of the total seven watersheds in the region, the selected water sites evaluated in this study were located in the four southernmost contiguous watersheds; the Schuylkill, Darby-Cobbs, Tacony-Frankford and Delaware basins.

Four prominent water-bodies with maximum recreational use were identified: Schuylkill River (site- Fairmount Dam (FD), site-Bartram Garden (BG)), Delaware River (site-Pleasant hill park (PH), site-Pennypack park (PP), site-Frankford Aresenal (FA)), Cobbs creek (site-Cobbs Education center (COB)), and Tacony creek (site-Adams avenue dam(AD), site-Bingham street (BH). The Schuylkill River flows predominantly through areas of industrial uses, while Delaware River flows through areas of mixed uses (commercial, industrial, dense residential and park lands). All the sites on both the rivers had prominent fishing locations; pier, balcony or dock. In addition, PH and FA also had a boat launch dock. Site COB was shallow (average water depth-0.68m, discharge-0.235m^3/sec, width – 14.4m), only accessibly by foot, and was located on a stream draining into Delaware River. Sites AD and BH are located along a fourth urban stream (Tacony creek), and are also typically very shallow (average water depth-0.69m, discharge-0.91 m^3/sec, width- 21m). Access to Site BH is very limited, due to its location on a riverbank densely populated with trees and uneven terrain. Site map and characteristics are presented in Figure 2.1 and Table 2.1 (respectively).

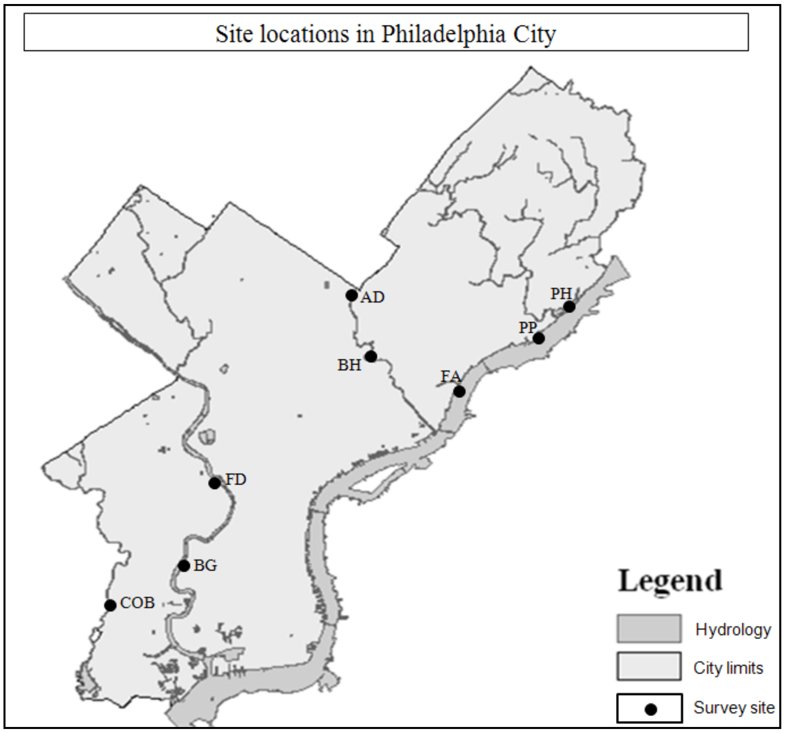

Figure 2-1 Site map of survey locations covered in Chapter 1.

Table 2-1 Site Characteristics

Waterbody		Schuylkill River		Delaware River			Cobbs Creek	Tacony Creek	
Site no.		FD	BG	PH	PP	FA	COB	AD	BH
Survey Year		2010	2010	2010	2010	2010	2008	2008, 2009	2008
Water body	Large urban river	x	x	x	x	x			
	Small shallow stream						x	x	x
	Access to water	x		x	x	x		x	
	Beach-front			x	x			x	
Zoning	Parklands	x	x	x	x	x	x		
	Residential	x	x	x				x	x
	Commercial				x		x		
	Industrial					x			
Accessibility	By foot -vegetated		x				x		x
	Walking trail	x		x	x	x		x	
	Distant car parking		x				x	x	x
	Adjacent car parking	x		x	x	x			
Infrastructure	Fishing pier	x	x	x	x	x			
	Boating ramp/dock			x		x			
	Sports/ park facilities	x			x				

2.2.2 Time-Lapse Cameras

A total of 11 HD MegaPixel IP cameras (Siteisight, 1125 Duchess Dr, Mclean, VA) were installed during the study period at 8 sites for data collection. The higher resolution (5 MP) cameras were selected as they allowed a more distant installation while still permitting discrimination between recreational activities. The cameras were mounted on trees, balcony or pole at heights ranging from 3-5m to improve security. Each unit was programmed to relay a live video stream via cellular broadband to a computer server, which stored the recorded pictures at two minute intervals. The camera assembly consisted of a timer, a cellular router, an external antenna, a wireless modem, a weather station, a thermostatically controlled cooling fan, 12V DC inverter and a weatherproof camera enclosure (heavy duty). The timer allowed the camera to run only during the day time (7am- 7pm) by automatically turning off the power supply to the system. At typical study locations that lacked a direct power line to operate the camera, DC voltage through deep cycle marine batteries was used, which were frequently changed (at 4 day intervals). In some locations, the batteries were charged through direct solar energy. Batteries were securely housed in aluminum alloy enclosures, and anchored to fixed structures at each site (i.e. tree or utility pole). Figure 2 shows examples of photos captured by time-lapse cameras at Sites FA and COB. The important design parameters of the camera assembly are outlined as following:-

- Camera- HD 5.0-MegaPixel IP camera
- Lens – varifocal lens 4.5-12 mm with manual iris & focus/zoom
- Power – powered from electricity pole (transformer 220V to 110V)
  - solar panels (2-day autonomy, 40watt solar module)
  - batteries (deep cycle marine MK powered AGM battery with 12V and 106Ah capacity)

- Data transmission: 55Mbps, up to 22fps
- Wireless Network: 3G/4G Sprint modem

Figure 2-2 Examples of pictures taken from the time-lapse cameras at sites PP and COB.

2.2.3 Remote Surveying

Images captured with the cameras and hosted on the password protected Siteisight server were remotely accessed using static IP and a Megaviewer Time-lapse service. The pictures were time, date and weather (temperature, pressure, dewpoint) stamped and organized chronologically. This service also offered zooming and image comparison features and allowed for multiple people to view images simultaneously on the Megaviewer. Archived images were scanned manually for any observable recreational use activity. The following data were collected and documented for all observations: activity location, date, day of the week, weather type (dry/wet), activity start time, activity end time, exposure type (swimming, wading, playing, boating, onshore fishing, fishing, jet skiing, kayaking), total users and miscellaneous notes.

2.2.4 Method Validation

In-person observational surveys of 3 hours duration were conducted one day per week between the hours of 9am and 6pm at all the water locations. Weather conditions, the number of the users, type of recreational activities and durations were recorded.

2.2.5 Statistical Methods

The student t-test was used to assess the difference between the number of recreators during wet and dry weather and during weekends and weekdays. Linear and polynomial fits were compared using a partial-F test for examining the relationship between the camera based observations and the in person survey logs for each water-body. The best distribution to model the exposure durations was determined via the KS goodness-of-fit test in Crystal Ball® (Statistical software) by using maximum likelihood estimation. Before generating the exposure duration statistics, a two sample student t- test was conducted in the statistical program "R" (www.r-project.org) to verify if the mean duration for a particular activity for wet and dry weather condition could not be assumed to be statistically significantly different.

2.3 Results

2.3.1 Time-lapse Camera Observations

Observed activities were grouped into two categories; primary contact activities and secondary contact activities. Primary contact activities are defined as those activities where the body could be fully immersed with a significant potential to swallow water. These activities included swimming, jet skiing, tubing, water skiing, windsurfing, kayaking and wading. Secondary contact activities include boating, boat fishing, fishing, playing with water, playing with wet dog, canoeing and ferry rides in which there is direct contact but a very low chance of swallowing water. Due to the comparatively similar extent of water-exposure found in water skiing, wind surfing, tubing and swimming, these were grouped together in one category, “swimming”.

Observations were grouped into dry weather and wet weather categories. In this study, wet weather included the day of a rain event (> 0.1 in), plus the following 72 hours. Based on independent water quality analyses reported by the municipality, 0.1 inch was the minimum amount of precipitation required to trigger a CSO event and bacterial concentrations remained elevated 72 hours after a rain event (data not shown).

Table 2.2 summarizes the influence of weather conditions on the recreational use pattern, detailing the average number of people observed conducting a particular recreational activity on days the activity was observed. The p-values less than 0.05 were deemed statistically significant correlations and were bolded in the table. Not all activities were observed everyday due to factors such as weather, day of the week, or camera malfunction. It was found that the effect of weather on the type of recreational activity observed varied by water location. For example, at sites PH and FA greater numbers of users were boating during dry weather compared to the number of boaters during wet weather ($t= 1.98, 2.62$; $p=0.03$, $p=0.01$). Similarly at site PP the average number of users playing with water during dry weather were greater than those seen during wet weather ($t= 2.13$, $p=0.04$). Likewise at site FA, greater number of users jet skied during dry weather, as compared to wet weather users (t-stats: 2.07, $p=0.04$). Interestingly, this pattern of dry weather preference in activity did not appear for the sites with moderate usage statistics (i.e. sites along Schuylkill river and at creeks sites) (Table 2.2, Figure 2.3). The reason for this discrepancy lies in the criteria used for defining the wet weather event. It should be noted here that our wet weather definition (72 hours after the onset of rainfall) is predominantly evaluating the exposure scenario related to impaired water quality due to a rain event. However, this definition of wet weather suppresses the real impact of a rainy day on the recreational usage by including no-rain days into the wet weather period.

Table 2-2 The effect of weather (wet versus dry) on recreational activity patterns.

Daily average number of the users/total users observed with camera at each location, by activity for dry and wet weather conditions.																	
Recreational Use activities		**FD**				**BG**				**PH**				**PP**			
		Dry[a] (n=94)	Wet (n=61)	t-stats	p-value	Dry (n=97)	Wet (n=66)	t-stats	p-value	Dry (n=89)	Wet (n=59)	t-stats	p-value	Dry (n=82)	Wet (n=58)	t-stats	p-value
Primary	Jetski	3/106	3/44	0.05	0.96	2/21	2/5	0.46	0.66	19/1098	15/407	0.85	0.40	2/19	2/21	0.041	0.96
	Kayaking	8/116	28/224	-1.36	0.21	2/15	1/1	1.83	0.11	1/22	2/8	-0.41	0.69	2/17	2/6	1.010	0.34
	Wading	2/7	0	1.75	0.18	-	-	-	-	3/10	3/11	0.37	0.73	6/381	5/212	0.57	0.57
	Swimming	-	-	-	-	-	-	-	-	6/212	9/60	-0.61	0.57	4/43	3/10	0.850	0.42
Secondary	Boating	9/629	8/261	0.65	0.52	7/350	6/139	0.86	0.39	143/12645	7/303	2.16	**0.03**	42/3155	41/1985	0.230	0.82
	Fishing	12/946	11/627	0.12	0.90	7/615	7/409	0.41	0.68	4/269	4/146	0.54	0.59	11/865	9/492	1.560	0.12
	Boat fishing	6/334	5/101	1.74	0.08	4/16	1/3	1.73	0.91	3/66	3/19	0.77	0.45	2/8	2/7	-1.400	0.21
	Playing	-	-	-	-	-	-	-	-	4/81	3/25	0.95	0.35	7/380	5/141	2.130	**0.04**
	Playing with dog	-	-	-	-	-	-	-	-	1/24	1/5	1.95	0.07	2/85	2/69	-0.570	0.57
Recreational Use activities		**FA**				**COB**				**AD**				**BH**			
		Dry (n=29)	Wet (n=26)	t-stats	p-value	Dry (n=50)	Wet (n=34)	t-stat	p-value	Dry (n=66)	Wet (n=64)	t-stat	p-value	Dry (n=15)	Wet (n=5)	t-stat	p-value
Primary	Jetski	9/165	5/53	2.070	**0.04**	-	-	-	-	-	-	-	-	-	-	-	-
	Kayaking	1/4	1/6	1.000	0.42	-	-	-	-	-	-	-	-	-	-	-	-
	Wading	-	-	-	-	3/104	3/49	0.6	0.55	3/97	3/57	0.723	0.47	5/63	1/4	4.27	**0.001**
	Swimming	3/16	3/3	0.090	0.93	-	-	-	-	-	3/10	-5	1.99	0	0	-	-
Secondary	Boating	31/1112	22/538	2.610	**0.01**	-	-	-	-	-	-	-	-	0	0	-	-
	Fishing	48/1249	45/1027	0.630	0.53	2/9	1/11	1.31	0.26	2/24	3/34	-0.42	0.678	4/4	0	9.46	**<0.0001**
	Boat fishing	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Playing	5/21	3/5	1.470	0.27	4/72	4/58	0.66	0.52	5/56	4/39	0.438	0.667	-	-	-	-
	Playing with dog	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

[a] "n" represents the total number of dry/wet weather days recreational use data was collected from the time-lapse camera at a given site.

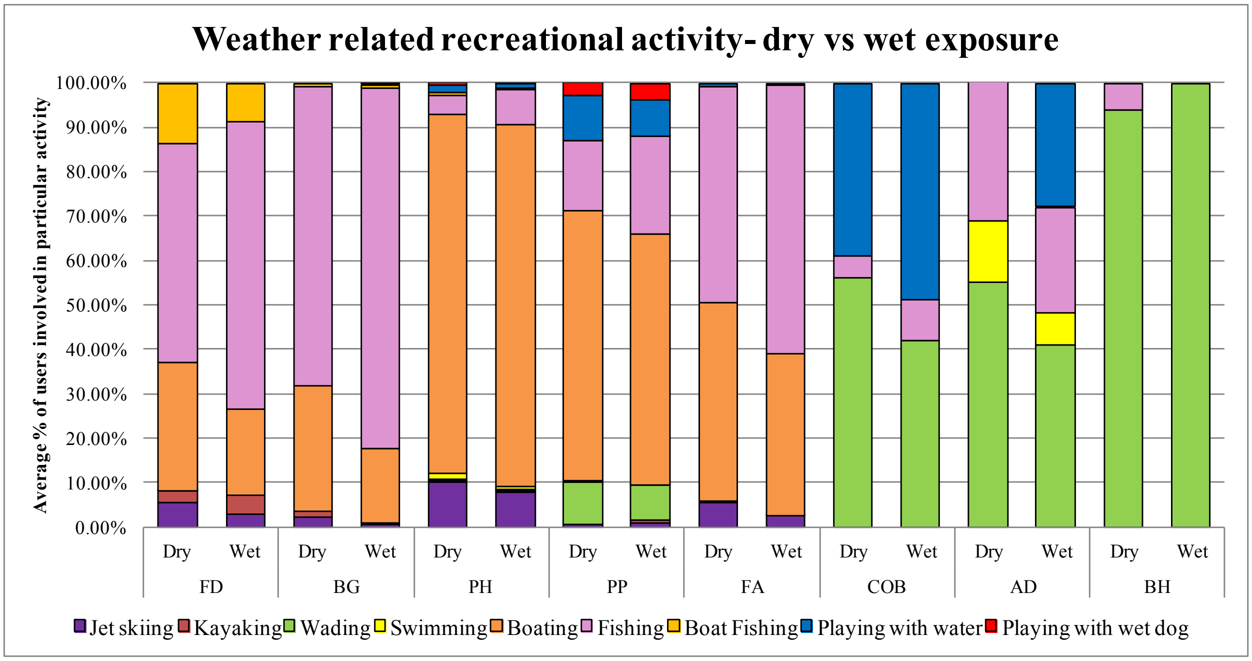

Figure 2-3 Users proportion distribution per activity during dry and wet weather.

Again, at several water sites, recreational patterns varied depending on whether the day was a weekend day or weekday (Table 2.3, Figure 2.4). For example, the influence of the type of the day was significant at site FD, with greater number of the users being involved in every observable activity on weekend, as compared to weekday users (Table 2.3). Also, it was found that more people opted to boat on weekends at both the rivers (Schuylkill and Delaware River) than boating on a weekday. Large number of recreators at site PH and site FA were jet skiing on weekend, but there was no statistical difference between the percentage of weekend and weekday jet skiers at Site FD and BG (Table 2.3, Figure 2.4). This observation of greater usage on weekends as compared to weekdays is consistent with other studies that suggest recreational activity impacts the bacteriological water quality according to the day of the week (Varness, Pacha et al. 1978; Graczyk, Sunderland et al. 2007). However, there were no differences in activity patterns regarding the day of the week at Sites COB and BH. These sites had limited access due to distal location from main road (Table 2.1), therefore attracting approximately same number of users during weekday and weekend.

The camera output information was further analyzed to develop exposure duration distributions using KS-goodness of fit test for 5 candidate probability distributions- log normal, weibull, gamma, exponential and beta distributions. In quantitative microbial risk assessment, the determination of number of people being exposed to pathogens via multiple pathways (boating, fishing, wading, playing etc) and the associated length of time spent on daily basis are essential exposure factors. Comprehensive information on these parameters for non-swimming recreational activities have not been quantified in peer reviewed literature and are presently not available. These would provide more precise estimates on ingested dose and was a factor in how we grouped exposures for evaluating the camera images. Table 2.4 and Table 2.5 details the measured durations of activities in which people were potentially exposed to water while partaking in various water related recreational experiences.

Table 2-3 Temporal effects (weekday versus weekend) on recreational activity patterns.

Daily average number of the users/total users observed with camera at each location, by activity for weekdays v/s weekends																	
		FD				BG				PH				PP			
Recreational Use Activity		Weekday n=109	Weekend n=46	t-stats	p-value	Weekday n=115	Weeken d n=48	t-stats	p-value	Weekday n=104	Weeken d n=44	t-stats	p-value	Weekday n=101	Weekend n=39	t-stats	p-value
Primary	Jetski	2/47	5/103	-3.64	**0.001**	1/5	2/21	-1.70	0.115	12/681	28/824	-3.26	**0.002**	2/11	3/29	-1.586	0.137
	Kayaking	3/20	19/319	-2.11	**0.050**	1/5	2/11	-1.22	0.276	2/24	2/6	0.29	0.782	2/13	3/10	-0.927	0.404
	Wading	3/6	1/1	*b*	*b*	-	-	-	-	2/9	4/12	-1.18	0.353	5/313	9/280	-3	**0.005**
	Swimming	-	-	-	-	-	-	-	-	7/195	6/77	0.96	0.342	3/22	5/32	-1.115	0.310
Secondary	Boating	6/337	14/553	-5.48	**<0.0001**	5/208	9/281	-2.29	**0.027**	77/7644	227/9890	-3.55	**0.001**	34/2749	67/2395	-4.425	**<0.0001**
	Fishing	10/978	15/595	-3.76	**0.0004**	6/624	9/400	-2.30	**0.024**	4/283	4/132	-0.19	0.849	9/800	14/557	-4.070	**<0.001**
	Boat fishing	5/227	7/208	-2.31	**0.025**	5/10	3/9	*b*	*b*	3/42	4/45	-0.20	0.070	2/6	2/9	0.291	0.786
	Playing	-	-	-	-	-	-			4/76	4/30	-0.13	0.898	5/247	9/274	-3.687	**0.001**
	Playing with	-	-	-	-	2/2	0	*b*	*b*	2/24	1/5	2.45	**0.027**	2/102	2/52	-1.208	0.235
		FA				COB				AD				BH			
Recreational Use Activity		Weekday n=37	Weekend n=18	t-stats	p-value	Weekday n=60	Weeken d n=24	t-stats	p-value	Weekday n=96	Weeken d n=34	t-stats	p-value	Weekday n=13	Weekend n=7	t-stats	p-value
Primary	Jetski	5/89	12/129	-2.71	**0.017**	-	-	-	-	-	-	-	-	-	-	-	-
	Kayaking	1/6	1/4	1.00	0.374	-	-	-	-	-	-	-	-	-	-	-	-
	Wading	-	-	-	-	3/109	3/44	0.45	0.653	3/79	4/75	-2.14	**0.040**	3/36	5/31	-0.61	0.561
	Swimming	1/1	4/18	*b*	*b*	-	-	-	-	0	3/10	-5.00	1.990	0	0	-	-
Secondary	Boating	26/930	48/868	-2.55	**0.017**	-	-	-	-	-	-	-	-	0	0	-	-
	Fishing	42/1390	56/892	-2.04	0.055	2/16	1/4	1.94	0.088	2/25	3/33	-0.42	0.678	4/4	0	1.34	0.180
	Boat fishing	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Playing	3/10	5/18	-0.61	0.574	4/91	4/39	0.06	0.954	4/40	4/55	0.01	0.990	-	-	-	-
	dog	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

[a] "n" represents the total number of weekend/weekday days recreational use data was collected from the time-lapse camera at a given site.

[b] The effect of type of day could not be evaluated statistically at these locations, because of lack of sufficient data.

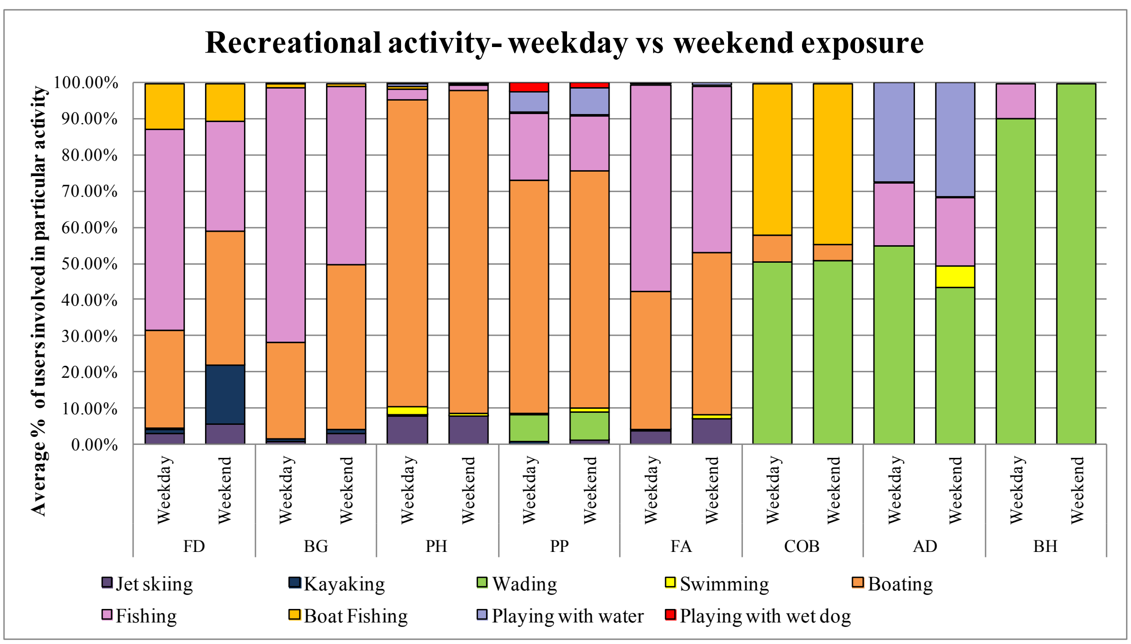

Figure 2-4 Users proportion distribution per activity during the weekends and weekdays.

Table 2-4 Exposure duration statistics for non-transient activities, by water body.

Water body	Activity	Duration (hrs:min)		
		Mean	Median	95 Percentile
Schuylkill River	Fishing	2:36	2:09	6:11
Delaware River	Fishing	2:07	1:49	4:49
	Wading	0:14	0:06	0:52
	Playing	0:17	0:11	0:53
Creeks	Fishing	0:31	0:24	1:58
	Wading	0:08	0:04	0:28
	Playing	0:22	0:08	1:22

Table 2-5 Exposure duration distributions for non-swimming recreational water- exposures based on camera observations.

Activities		Site FD[a]		Site BG[a]		Site PH[b]		Site PP[b]		Site FA[b]		Creeks[c]	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Transient	Jet skiing	Tri(1,2,6)				LN(0.31,0.52)		Wei (0.06,1.56)		B(0.03, 0.43, 0.56, 1.43)		N/A	
	Boating	Tri(2,3,6)				Wei(1.35,1.06)	LN(0.8,1.45)	Wei(0.45,1.19)		B(0.03,1.5, 1.61,6.21)	B(0.03, 1.0,1.11,2.48)	N/A	
Non-transient	Wading	N/A				LN(0.25,0.62)				N/A		LN(0.132, 0.145)	
	Fishing	G(1.3, 2.0)				LN(2.11, 1.35)				G(1.16,1.57)	G(1.05, 1.5)	G(0.44, 1.16)	
	Playing with water	N/A				LN (0.28, 0.6)				N/A		LN(0.368, 0.642)	

[a] Duration of transient activities is based on the professional judgments provided by the Schuylkill River Development Council personnel.
[b] Duration of transient activities represents the person hour of use of a particular activity at a site.
[c] Duration data pooled for sites along the creeks, because at 0.05 level of significance the duration statistics for observed activities did not differ significantly between these sites (Site 3A, 4A and 4B).
[d] Tri-Triangular(min,mode,max); G-Gamma(scale,shape); LN-Log normal(mean, SD); B-Beta(min,max,alpha,beta); Wei-Weibull(scale,shape)

A two sample student t-test was conducted to determine whether the duration data for a particular non-transient activity from each site were different. As a result, the duration of activities observed at site COB, site AD and site BH were pooled to develop one combined distribution for each activity. Similarly, the duration of non-transient activities at site PH and PP were pooled to represent one common distribution for each activity. However, the fishing data at sites along the shallow creeks and those along the large rivers were not pooled because of the statistically significant difference between the average durations spent fishing at these two different water-bodies.

Due to immobility of the cameras, the duration of certain mobile transient activities (boating, kayaking, canoeing and jet skiing) were calculated differently than the sum of time recorded on film used for non-transient activities (wading, playing and fishing). The non-transient duration estimates based on camera observations (for the mean, median and 95 percentile by activity) for each water-body are summarized in Table 2.4. It was found that on an average an individual would spend 75% more time fishing at a river as compared to fishing at a shallow water body (Table 4; river=2hrs 7min, creek=31min.). Playing and wading refers to the activity at the beach front along the water's edge; playing involves splashing and throwing rocks in the water, while wading describes the recreator walking with their feet in the water. Wading duration was longer at beach fronts along the river than at creeks (43%), while playing duration was longer at creeks than at rivers (22%) (Table 2.4).

Additionally, the activity duration distributions were developed for both transient and non-transient activities (Table 2.5). Based on the professional judgments provided by the river development council personnel, it was found that the minimum and maximum time spent by people boating and jet skiing in Schuylkill River ranged from 1 to 6 hours and 2 to 6 hours respectively. Because there was limited data for trip duration but it was assumed to have a continuous probability distribution, a triangular distribution function was selected to be an appropriate model as it is based on a minimum, maximum and mode and can have a variable symmetric to asymmetric density function (Table2.4). Triangular distributions are appropriate when the most likely outcome is known (Haas, Rose et al. 1999). Due to the presence of dedicated boat launch docks at sites along Delaware River, we were able to analyze the camera images to account for the person hour of use of particular transient activity at the sites on this river. Since the person hour of use represents the sum of time frames for which a particular activity was seen at a site, the duration record for transient activities at Delaware River had variation across the dataset. As a result, Beta distribution provided the best fit at site FA, while Weibull and Log normal was found to best describe the durations at sites' PH and PP (Table 2.5).

The Beta distribution is a versatile way to represent outcomes which are constrained within an interval defined by a minimum and maximum value. The Log normal and Weibull distributions best describe data sets containing extreme values with moderate to severe positive skewness, as observed at site PH and site PP. For non-transient activities, the Log normal characterized the wading and playing duration dataset, while the Gamma distribution fitted best to the fishing duration record for all the sites. The goodness of fit for all the distributions was determined using a Kolmogorov-Smirnov (K-S) test. From this test, the distribution which returned the lowest K-S value provided the best fit.

2.3.2 Validation of Remote Time-lapse Camera

All eight sites were surveyed for 3 hours alternately once a week during the study period of 2008 – 2010 for validation of camera based observations. The camera images were compared with the associated in-person survey logs to check for discrepancies in the collected data. Since the average number of users observed per day based on both survey techniques at sites along Delaware River and along the creeks were not significantly different, the data was pooled. However, observations of site FD and site BG were found to represent two statistically independent groups of recreators, so the data were treated separately while comparing the camera and in-person survey logs.

In-person data were graphed with camera interpretation results to propose possible correlations between the two. Proposed linear and quadratic relationships were reviewed as to their applicability using correlation factors as the discriminator. Some R^2 correlation factor terms between the modeled quadratic function and the data were higher than that derived using a linear function. As a result of the partial F test on the significance of the quadratic term when comparing the in-person observations with the camera data, it was found that the improvement in the R^2value was not significant over the linear term (data not shown). Hence, linear fits were chosen to represent the correlation between the two observation techniques (Figure 2.5 a, b).

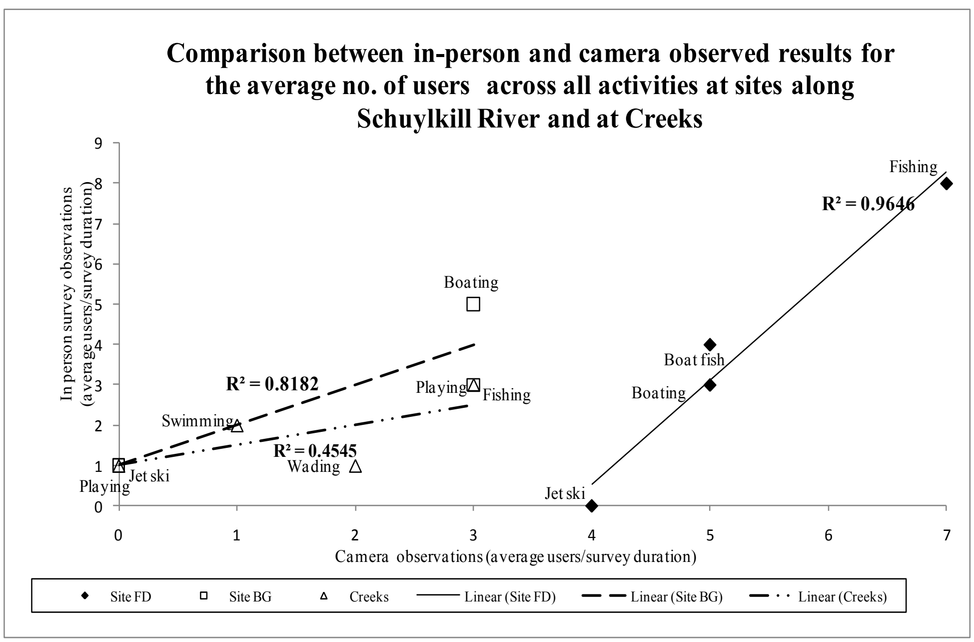

Figure 2-5 (a) Comparison between average number of recreators as confirmed by direct observation versus camera surveillance for Schuylkill River sites.

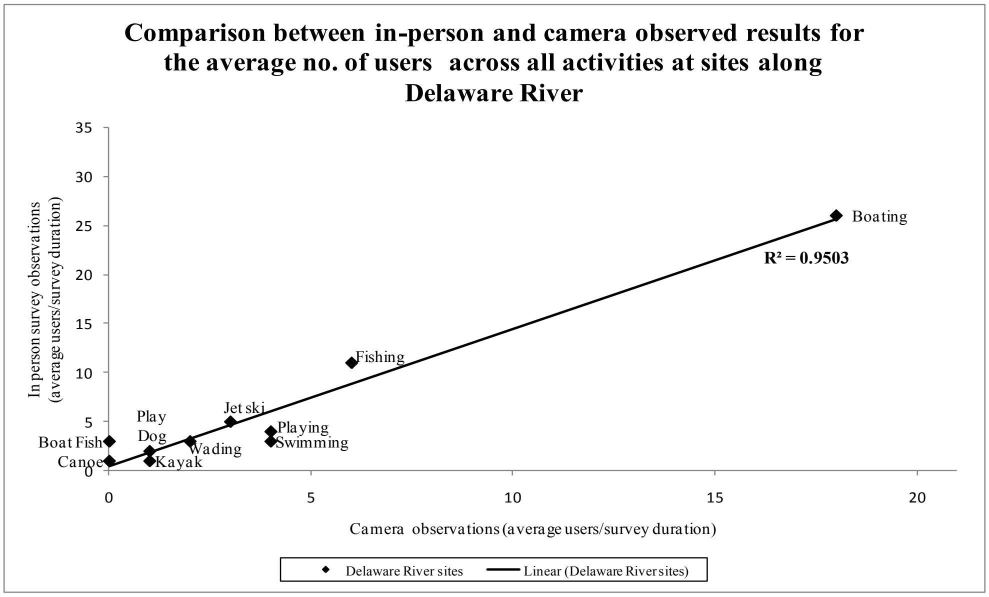

Figure 2-6 (b) Comparison between average number of recreators as confirmed by direct observation versus camera surveillance for Delaware River sites.

The resulting regression model fit the sample data very well, producing high R^2 values- r^2=0.9646, 0.8182, 0.9503; for site FD, site BG, and sites along Delaware River (respectively). Lower R square value for sites along the creek (r2=0.4545) can be explained due to relatively small range of observations (0-3 users/day). By comparing the slopes of each linear model, we found that the users count based on in person surveys was greater than the corresponding camera output summary at all sites except for the creek sites. This is expected as the field of view for the in person survey was wider than the area captured by the camera. Also, some activities (especially transient activities) could have been missed due to the interval time between images (2 minutes) resulting in slightly lower counts of users based on the camera surveys. An exception to this was evident for the creek sites' comparison graphs where the linear fit had a flatter slope than a 1:1 regression line. This discrepancy of higher users recorded by camera images as compared to the in-person surveys is associated with the underlying assumption related to image interpretation at these shallow water bodies. It was assumed that recreators standing close to the water edge would have entered the water in the intervening time between the recorded images, and so the number of water-exposed users was overestimated. However, in the corresponding in-person surveys, only actual exposed users were counted.

While the analysis substantiates that the remote camera captures a representative subset of the entire phenomenon, it should be noted that camera (even with adjustments) loses some information in the interval between the images and cannot capture the entire field of view, so will not produce a 1:1 linear correlation. Thus any discrepancies have to be examined for bias, along with reasonable adjustments for addressing these shortcomings.

2.4 Discussion

This study describes a novel method for data collection of recreational use patterns on large and small urban streams. Results of this study indicate that non-swimming recreational use patterns vary temporally and spatially, the prevalence of particular activities at a given location was primarily determined by two factors: geography (including ease of access and water depth) and surrounding development. In several sites, significant differences in use patterns were observed during weekends compared to weekdays, as well as during wet and dry weather. However, these temporal differences due to weather or day of the week, were not uniform across study sites, highlighting the importance of understanding specific recreational patterns and of collecting comprehensive data when conducting a risk assessment on a watershed.

The key to calculating the most accurate exposure dose is to identify users' distribution and the exposure duration values that specifically relate to the exposure situation being assessed. With the help of this novel technology, we were able to collect comprehensive data on these factors.

With the water quality data and the recommended ingestion rate estimates, precise ingested dose for particular recreational experiences can be developed.

In future studies, time-lapse cameras could additionally be used to identify the number of hand to mouth occurrences per hour for a given activity or the duration of swimming with the head above or below water, which could subsequently inform more accurate recreational use type-specific ingestion volumes. To collect data for estimating hand-to-mouth transfer (e.g., during fishing), cameras would likely need to be operated at much higher frame collection rates or in a streaming mode. As camera and broadband technologies evolve, the opportunities for simultaneous, temporal and spatial data collection with greater rates of frame capture, expands for waterborne exposure risk assessments.

In our study, we employed eleven high resolution (HD, 5MP) time-lapse cameras on eight sites, based on the location and viewing range desired. At each of the three sites (site FD, site BG and site FA) we used two cameras due to distributed opportunities of recreational use. High resolution cameras permitted camera installation at a reasonably far distance from the target area, which helped avoid vandalism. The higher resolution cameras had sufficient image clarity to permit the identification of number of recreational users, and recreational use type.

The pitfalls of relying upon remote camera surveillance for research have been echoed by many investigators and highlight the importance of conducting field verification of the data collection. The occurrence sites selected for observation depend on the assumption that the detection captures most of the activity at a site (as in this case), or that the activity is equally distributed in the geographic area and equivalent between individuals. Furthermore, sufficient detail must be provided by the cameras to distinguish between individuals and their specific activities so as to rule out repetitions and note significant differences(Cutler and Don 1999). In this study, limitations of the cameras' field of view and the time gaps between film shots posed problems in tracking and differentiating distant boaters and jet skiers. Computing the time interval of each leisure activity and distinguishing between different groups of boaters was difficult. Use of remote surveillance is more complex than in-person surveys, requires weekly battery changes, and necessitates occasional troubleshooting. Additionally, some data can be missed. For example, time-lapse camera shots are periodic and may not capture every action, depending on the designated picture interval. However, in-person surveys are labor intensive, may deter recreators, be prohibitive in some unsafe areas, or require multiple personnel. Unlike in-person surveys, cameras can operate continuously, capturing more observational data overall. Relying on results from solely periodic in-person surveys can skew the observational data because it captures only a small fraction of the potential observation time in a recreational season.

However, the advantage of saving time, money and effort through using remote sensing over direct observation outweighed the drawbacks. Multiple sites were recorded in a consistent, unbiased manner with data records that could be reviewed, interpreted and verified at a later date.

The estimated cost savings based merely on a person-hour basis was determined by the differences between doing in-person surveys as compared to the camera analyses. Since the cameras recorded 12 hours/day/site, the in-person surveys would require an equivalent cumulative 84 hours/site/week. In comparison, battery/equipment maintenance and video interpretation for the camera surveillance totaled an estimated 12-14 man-hours/site/week Depending upon the type of power supply available at each site, the equipment expenses for conducting the camera survey totaled approximately $4,500/camera for site FD (with a direct power supply), $5,800/camera for the sites using deep marine cycle batteries (site BG, PP, COB, AD, and BH), $7,000/camera for site PH (including costs for constructing a trench to carry the power line from a utility pole to the camera location) and $10,200/camera for site FA (including costs for solar panels and installation charges). The basic equipment cost included camera(s), a 6 month subscription charge for image hosting at a server, and batteries with an enclosure box and connection accessories. With the expense of one camera setup per site added, the total expenses of additional equipment and labor for camera surveillance for one site/ week equaled approximately 45% of the entire expenses incurred using in-person surveys only for assessment.

2.5 Conclusions

Time-lapse cameras with periodic in-person validation allow for a rapid collection of recreational exposure and activity pattern data. This scheme requires fewer personnel than a full in-person survey program, provides potentially more reliable data than in-person surveys alone, yields an archive of recreational exposure data for future reanalysis, allows for real-time data collection with validation, and provides exposure-specific information, which can be applied to a variety of exposure assessment scenarios. This method has its own limitations due to recording gaps between time intervals of camera images as well as a limited coverage area according to its adjusted field of view. But, by conducting sporadic in-person surveys, the issues arising from limited camera views can be assessed. With the quick pace of advancement in camera technology and broadband availability, images with a higher frequency will be accessible in the near future. Incorporating discrepancies from comparative camera and in-person survey data can improve an overall risk assessment, ultimately affecting management decisions for a region or watershed.

The resulting recreational use data from this study, if collected in conjunction with fecal indicator bacterial water concentrations, could produce more accurate estimates of risk than those based on guessed exposure frequencies, recreation durations, or on mismatched exposure and water quality data. This novel method offers risk managers, urban planners and beach officials a powerful tool to conduct exposure assessment by using a cutting edge “time lapse” camera technology for collecting real time human water-exposure data. The detailed information on how the water are being used (swimming, fishing, boating, wading etc.) and how the usage and quality varies with wet weather conditions would help decision makers optimize management of their waters resources and identify the best use of resources for a particular water body.

3 Chapter 3: Quantitative Microbial Risk Assessment for recreational exposure to water bodies in Philadelphia, based on *Enterococci* model

3.1 INTRODUCTION

Philadelphia is one of America's earliest-developed cities and thus comes with an aging water and wastewater system. The City has three water treatment plants, three wastewater treatment plants, and 164 combined sewer-storm water outfalls (CSO)s that total more than 3,000 miles of water mains serving 1.5 million residents (Office of Watersheds) . The city's combined sewers are currently designed so that during heavy storm events the untreated wastewater is diverted to waterways instead of being treated in one of the three treatment plants. These discharges may exceed the water quality standards designated for optimal use of the water body. Any recreational exposure to this water during periods of increased pathogen concentration has the potential to cause illness. As such, the Philadelphia Water Department (PWD) has recommended that when a combined sewer outfall is overflowing, and up to a period of 24 hours following the rainfall event, residents should not recreate in the water body due to possible pollutant contamination. Moreover, the City does not permit swimming and bathing in any of the rivers and streams outside of organized events (e.g., races, triathlons, etc.) due to elevated risks of drowning from strong currents, injury from submerged objects, commercial shipping (in the Delaware) and other hazards.

Although swimming in Philadelphia waters is prohibited, other various primary and secondary water contact recreational activities are permitted such as jet skiing, fishing, boating, kayaking, tubing, canoeing and playing with water and have been surveyed in a previous study (Sunger, Teske et al. 2012). These non-swimming recreational activities have comparatively less exposure potential but they still need to be analyzed in order to quantify the depress of risk that such exposures confer. Most contact studies report swimming-associated health risks and neglect incorporating the potential of non-swimming exposure statistics in their estimates (Kay, Fleisher et al. 1994; Soller, Schoen et al. 2010). A recent study by Rijal and co-authors on Chicago Area Waterways (CAW) addressed this issue by evaluating health risks associated with incidental contact water-recreation (Rijal, Tolson et al. 2011). Only three incidental contact activities (canoeing, fishing and boating) were considered in this study with major focus on the effect of wastewater disinfection before it discharges into the CAW (Rijal, Tolson et al. 2011).

Runoff and wet weather discharges have been implicated in recreational water illnesses since the 1970's. One study (Curriero, Patz et al. 2001) used waterborne outbreak reports (between 1948 and 1994) to establish a strong statistical association between disease outbreaks and high total monthly precipitation using the Fisher exact method. A detailed risk analysis conducted for the Lower Passaic River (New Jersey) by other researchers (Donovan, Unice et al. 2008) estimated higher annual risk of gastrointestinal (GI) illness for homeless persons (0.88/year) as compared to that of visitors and recreators (0.14 - 0.7/year), by investigating a series of pathogens associated with CSO discharges into the river. The risk of infection from *Cryptosporidium* and *Giardia* per exposure event was reported to be 0.0035% and 0.101%, respectively, for occupational divers professionally exposed to fecally contaminated canal water in Amsterdam (Schets, van Wijnen et al. 2008). . Another study (Kay, Crowther et al. 2008) assessed the effect of CSO's on surface water bodies in the United Kingdom (UK) by quantifying fecal indicator organism concentrations and export coefficients for catchment with different land use patterns. Urban areas were identified as one of the key sources of fecal indicator organisms, with significantly higher values occurring for high-flow conditions, during or after rainfall.

Presently, the statewide water-uses identified under Chapter 93 of the Pennsylvania Code (The Pennsylvania) characterizes surface water recreation in three categories – boating, fishing and water contact sports (§ 93.4). Specific water quality criterion (§ 93.7) is recommended only for water contact sports, which refers to swimming and related activities. This standard relies upon fecal coliforms as indicator organisms with a maximum permissible concentration of 200 coliforms per 100 milliliters (geometric mean of 5 consecutive samples during 30 day period). Though the criterion is protective of water quality, it does not reflect the specific risk of illness posed by recreational use of a particular water body. A different approach to judge if the waters are safe for recreation would translate the bacterial water quality into the associated health risks posed by humans upon exposure to such waters. This approach is based on epidemiological studies and results in a dose-response relationship. Using such mathematical models, it can be correctly identified if the city water will pose any health risk to its users, and if so, to what extant will that be associated with wet weather CSO discharges.

Many epidemiological studies have demonstrated good correlation between the density of bacterial indicators (especially *Enterococci* and *E. coli*) and human illness resulting from swimming in waters contaminated with point sources of pollution (Wade, Pai et al. 2003; Wade, Calderon et al. 2006). Where non-point sources are dominated by fecal input, some researchers (Colford, Wade et al. 2007) have argued there is substantial uncertainty in the use of traditional fecal indicators in predicting illness. There is a vast recognition that the current standards do not adequately account for different usages of recreational waters, and that with new detection methods it is possible to test additional microbial agents more rapidly. To address these increasing concerns about the applicability and efficiency of recreational water quality criteria for fecal indicators, a revision methodology draft is recently provided by U.S. EPA (Office of Water 2012). In this draft, U.S. EPA has proposed a qPCR methodology as a rapid analytical technique for detection of *enterococci* in recreational waters. But due to insufficient information about the performance of this method under varied water-body conditions and the limited experience of its use in the field, EPA encourages a site-specific assessment of the method's performance before it is adopted into State Water Quality Standards. However, enterococcus is still recommended by U.S. EPA as a useful indicator of health risk in both salt water and fresh water. The current EPA water quality criteria model was developed using a prospective-cohort epidemiological investigation of people swimming in waters known to be contaminated with fecal pollution. It relates excess HCGI illnesses (Highly Credible Gastrointestinal Illness) to *Enterococcus* concentration and recommends the advisory limits for freshwater recreation as 61 colony-forming units (CFU)/100ml for single-sample maxima and 33 CFU/100ml for the 5-day geometric mean maxima, with the associated additional risk of 0.8% for recreational swimmers (Cabelli, Dufour et al. 1982; Dufour 1984). The HCGI case definition as recommended by U.S. EPA is vomiting, diarrhea with fever or a disabling condition and stomachache or nausea accompanied by fever.

Epidemiological studies have shown a number of adverse health outcomes (including gastrointestinal, dermatitis and respiratory infections) associated with exposure to recreational waters. The most recent national surveillance report of waterborne disease outbreaks by Center for Disease Control and Prevention (CDC) reported 78 recreational water outbreaks from 2005-2006 (Yoder, Hlavsa et al. 2008). In the report, ingestion was recognized as the most common source of exposure (61.5 percent) with the majority of outbreaks due to gastroenteritis (61.5 percent). Other illnesses related to recreational water exposure include otitis, conjunctivitis (Dewailly, Poirier et al. 1986; Yau, Wade et al. 2009), swimmer's itch (Verbrugge, Rainey et al. 2004), hepatitis A (Mahoney, Farley et al. 1992), Leptospirosis (Narita, Fujitani et al. 2005), meningoencephalitis (Heggie 2010), and Pontiac fever (Mangione, Remis et al. 1985; Modi, Gardner et al. 2008).

The Philadelphia Water Department (PWD) has enacted a balanced "Land-Water-Infrastructure" approach to achieve its CSO control goals. As a unique CSO control program the Department has committed to implement a Green City, Clean Water plan which uses green stormwater infrastructure to reduce CSOs (Philadelphia Water Department 2011). Though, the department has been making massive investment for implementing alternate natural mechanisms such as infiltration, detention basins and evaporation for protecting region's waterway systems from stormwater runoffs, currently there has been no study to evaluate the existing waterborne public health status of these waterway users. Quantitative evidence of actual human health impacts caused by water-recreation during wet weather periods is critical in 1) evaluating the risk factors for disease exposure, and 2) identifying and implementing disease control measures by public health officials. Thus, a microbial risk assessment study was conducted in order to provide scientific insights from detailed analysis of recreational exposure elements leading to incidence of gastrointestinal illnesses in the recreators and present the conclusions to human health policy makers. The key endpoint of health outcomes in the present study is the incidence of Highly Credible Gastroenteritis Illness (HCGI) because not only does the vast majority of research to date in the field of recreational water quality and health focus on gastroenteric outcomes, but the current U.S. EPA water quality criteria based on 1986 epidemiological studies relates to these illnesses. It is important to highlight that the revised 2012 Recreational Water Quality Criteria (RWQC), uses a different definition of GI illness referred as NGI, which includes vomiting, nausea and stomachache without the requirement of fever. This case definition is broader than HCGI as it would include cases of viral gastroenteritis, which are not always present with fever. With this revised approach the EPA calculated a translation factor of 4.5 NGI per HCGI and reported the equivalent illness level of 36 NGI per 1000 recreators. The illness level associated with the 2012 RWQC water quality recommendations is approximately 6 to 8 cases of HCGI per 1,000 recreators in both fresh and marine waters, when the HCGI illness matrix is used instead of NGI.

The key to calculating the most accurate exposure dose is to identify the users' use distribution and the exposure duration values that specifically relate to the exposure situation being assessed. With the help of remote surveillance, we were able to collect comprehensive data on these factors for recreational sites that had great usage potentials and were located in proximity to an existing CSO. The results obtained from this survey are presented in our previous paper (Sunger, Teske et al. 2012) and were directly applied in this study to estimate health risk to the users. The main objective of the study was to generate quantitative information about the estimated human health hazard (HCGI illnesses) due to recreation in the surface water bodies within the city limits. The major focus of the study was to investigate the influence of wet weather discharges on health impacts of recreational users. A second objective was to identify the exposure scenario leading to the potential of maximum health risk to the users under the current usage profile of the waters. The study quantified the daily risk of HCGI illnesses using probabilistic techniques and presents the results under two categories; risk per 1000 users and total risk per observed number of users, using two different dose-response relationships for enterococci as an indicator organism representing the city's surface water quality.

3.2 MATERIAL AND METHODS

3.2.1 Study approach

A Quantitative Microbial Risk Assessment (QMRA) based approach was adopted to predict estimated risk of GI illnesses in Philadelphia from ingestion of recreational water that is subject to the wet weather discharges during the peak recreational season of the year. The risk was calculated during dry and wet weather conditions for sites located along the Delaware River, the Schuylkill River, Tacony Creek and Cobbs creek. QMRA is a technique that gives a direct and scientific framework for assessing water quality and determining whether the waters pose any health hazard to its users (Haas, Rose et al. 1999). A probabilistic framework for assessing the risk of illnesses was adopted for conducting this QMRA. Utilizing the probability distributions, we aimed to account for the variability and uncertainty that existed in our input parameters. To manage risks, it is important to identify which pathogen(s) and exposure scenario(s) lead to waterborne illness. Multiple exposure routes (ingestion, inhalation, or dermal contact) and scenarios (swimming and non-swimming) exist for water related recreational use. Incidental ingestion may occur through inhalation and entrapment of mists in the nose and mouth with subsequent swallowing. Typically, very less information is available for quantifying the amount of water that users in low-contact water use (such as canoeing, boating and playing with water) may ingest. Hence in a few cases, when the literature information was not sufficient to develop ingestion rate estimates, additional assumptions were made following the EPA exposure factor guidelines. Furthermore, dermal contact was ignored in the exposure analysis as it is assumed that exposure does not contribute to gastrointestinal illness.

In order to ascertain the weather influence on health outcomes, the collected information on recreation and water quality were divided under two main domains – dry weather data and wet weather data. The wet weather data included the observations collected during a rainfall of significant amount that may lead to CSO discharge. Separate ingestion estimates for all observed activities were generated to be applied in the GI risk calculation.

The observed proportion of different activities on a particular site was expressed on the basis of 1000 users. An alternative dose response model was applied to calculate the expected illnesses from water recreation and results were compared with the recommended 1986-criteria model (Dufour 1984). Eventually, a distribution of risk estimates was developed that described the range of illnesses that could be expected on a randomly chosen day at a recreational site based on the relative frequency of bacterial concentration, exposure duration, user's proportion by activity and ingestion volumes.

3.2.2 Site Description

Ten sites located close to an existing CSO outfall within the city of Philadelphia with a significantly high usage potential were monitored using time-lapse cameras during the peak recreational season (May through September) in 2008 to 2010. Three out of ten sites are located on shallow creeks; Tacony creek (site: Adams Ave Dam (AD), site: Bingham street (BH)), Cobbs creek (site: Cobbs Education Center (COB)). The remaining sites are located on large rivers: Schuylkill River (site: Fairmount Dam (FD); site: Bartram's Garden (BG)) and Delaware River (site: Pleasant Hill Park (PH), site: PennyPack Park (PP), site: Frankford Arsenal (FA), site: Penn Treaty Park (PT) and site: Penns Landing (PL)). Site characteristics, including the geographical and physical features have been discussed in detail in our previous publication (Sunger, Teske et al. 2012), except for site PT and site PL. In brief, all the sites on both the rivers had prominent fishing locations and a few sites also had boat launch docks. Sites at the creeks were shallow and accessible by foot.

Table 3-1 Site Characteristics

Waterbody		Delaware River	
Site no.		PT	PL
Survey Year		2010	2010
Water bod	Large urban river	x	x
	Access to water	x	x
	Beach-front	x	
Zoning	Parklands	x	
	Residential	x	
	Commercial		x
Accessibility	By foot -vegetated		
	Walking trail	x	
	Distant car parking		x
	Adjacent car parking	x	
Infrastruct	Fishing pier		x
	Boating ramp/dock		x
	Sports/ park facilities		x

The additional sites PT and PL both have easy access with dedicated parking and allow opportunities for fishing. Site PT was almost 5 river-miles downstream of Site FA with no specific infrastructure for recreational usage such as a dock or pier. In contrast, site PL (1.5 river-miles downstream of site PT) is a famous historical waterfront property of the City with lots of recreational features such as marinas, a yacht club and duck boat tours (amphibious boats that can accommondate about 30 people). Site characteristics of site PT and site PL are summarized in Table 3.1.

3.2.3 Data Sources

Exposure estimates: In order to determine the risk associated with recreational water use, the data acquisition process required information for five input parameters: 1) indicator bacterial concentration (CFU/100ml), 2) ingestion rates (ml/hr), 3) exposure duration (hr/day), 4) total users (users/day), and 5) proportion of users participating in each activity. Bacteriological data during 2007 and 2010 was acquired from the Bureau of Laboratory Services (BLS), a state-of-the-art laboratory performing the services for the Philadelphia Water Department (PWD). Due to the relatively drier summer in 2010, it was not possible to get a good data set on wet-weather water quality. Hence in summer 2011, additional sampling and analyses of water quality was carried out at Drexel University. Sampling procedures as described in detail in the USEPA microbiology methods manual (Bordner et al., 1978, Section II, A) were strictly followed during data collection. Briefly, samples were collected in sterile containers and stored on ice until analyzed. Samples were analyzed within 5 hrs from the start of the collection procedure, and analyses were completed within 8 hrs after collection of the samples. Enterococci test method-1600 as recommended by USEPA (Method 1600: mEI, USEPA 2002) was used for enumeration of enterococci in collected samples. Using a membrane filter (MF) procedure, direct counts of enterococci in the water based on the number of colonies that develop on the surface of a membrane filter were obtained. The membrane filter containing the bacterial cells were placed on mEI Agar and incubated for 24 h at 41±0.5°C. All colonies with a blue halo, regardless of colony color, were recorded as enterococci colonies.

The daily precipitation data were recorded from the National Climate Data Center (NCDC) for Philadelphia International Airport and applied uniformly to all study sites. Days when precipitation was reported greater than 0.1 inch, plus 72 hours after the rain event, were considered as a wet weather period. Based on independent water quality analyses reported by the municipality, 0.1 inch was the minimum amount of precipitation required to trigger a CSO event and bacterial concentrations remained elevated 72 hours after a rain event (data not shown).

The user's count, proportional distribution of user's frequency by activity and exposure duration statistics at each site were obtained from in-person surveys and time-lapse camera surveillance. As mentioned earlier, the survey results for all sites (except PT and PL) have been discussed previously (Sunger, Teske et al. 2012). The final ingestion estimates including the minor contribution from incidental ingestion for low-contact water sports as discussed in the CAWS study (Geosyntec 2008) were used to generate the exposure distributions for secondary contact water activities. Also, additional guidelines from EPA document on human health risk assessment were used for further exposure scenario descriptions (USEPA, 2000). Table 3.2 summarizes the ingestion rate estimates as derived from literature to conduct the risk assessment.

Table 3-2 Summary of ingestion rates estimates as obtained from literature

Observed Activity	Fitted Distribution (Mean, SD) Ingestion Rate (ml/hr)	Distribution Source
Jet skiing , Canoeing, Kayaking	LN(5,5) + fixed intake (4 ml/hr)[a]	Geosyntec, 2008
Boating: power boat, water taxi, tour boat, Ferry	LN(1,0.5)+ fixed intake (1ml/hr)	Geosyntec, 2008
Fishing (on and off shore)	LN(3,2) + fixed intake (1ml/hr)	Geosyntec, 2008
Wading	LN(10,5)[b]	EPA, 2000
Playing with Water	LN(12,6)[c]	EPA, 2000
Swimming, tubing, water skiing	Adults: N(25,5)	Dufour et al., 2006

[a]The fixed ingestion rate accounts for inhalation via mist and droplets in nose and mouth
[b]EPA (2000) recommends an average IR of 10ml/hr for adults, we assumed an SD of 5 ml/hr to create the distribution.
[c]EPA (2000) recommends an average IR of 10ml/hr for wading. We assumed an additional 2ml/hr for playing in the water and an SD of 6 ml/hr to create the distribution.

Dose Response (DR) estimates: Current geometric mean bacteria density guidelines of U.S. EPA were derived from the risk equation developed by Dufour in 1982. Dufour applied regression analysis on the epidemiological study data collected for both fresh and marine water beaches to fit a log linear model (equation 1) representing the relationship between concentrations of *E. coli*. or *Enterococci* in the water and the additional risk of gastrointestinal illness in swimmers (Cabelli, Dufour et al. 1982). Note that there is a threshold enterococci concentration (approximately 4.65 CFU/100 mL) below which no excess risk of HCGI illnesses per 1000 users above the background is projected.

The traditional interpretation of dose response data was to assume that there is no effect below some exposure level. This approach considers the presence of a threshold dose that is required to be ingested in order for microorganisms/chemicals to produce infection or disease. However, this approach is contradictory to conventional QMRA, where dose-response functions are non-threshold in nature. Hence, we developed alternative models to the 1986-criteria model consistent with commonly used microbial dose-response models. The same dataset related to enterococci concentrations was analyzed using the method of maximum likelihood estimation. The chi-square goodness of fit test was conducted to evaluate the best-fit model following the approach described in Haas et al. (1999). The exponential (Equation 2) and beta-poison (Equation 3) models were selected as probability models to develop the dose response relationship as they have been widely used in QMRA (Haas et al., 1999). The results were compared with the 1986-criteria linear (equation 4) model. In the case here, we assume an underlying baseline response that is dose-independent, upon which the dose-dependent response is superimposed. All the statistical analysis was done using "R", a statistical program and applying the Nelder–Mead algorithm. Both the candidate distributions assume that the organisms are Poisson distributed in the source-samples and that the probability of illness is binomial/beta distributed (Haas, Rose et al. 1999; Teunis, Nagelkerke et al. 1999; Teunis and Havelaar 2000).

$$P(d) = \frac{9.4.log10(d) - 6.278}{1000} \qquad \text{Eq (1)}$$

$$P(d) = a + (1 - a).(1 - e^{-k.d}) \qquad \text{Eq (2)}$$

$$P(d) = a + (1 - a).\left[1 - \left\{1 + \left(\frac{d}{N_{50}}\right).\left(2^{1/\alpha} - 1\right)\right\}^{-\alpha}\right] \qquad \text{Eq (3)}$$

$$P(d) = m.log10(d) + c \qquad \text{Eq(4)}$$

where P(d) is the probability of response at dose "d" and "k" is the probability that a single organism can survive and initiate infection. "N50" is the median infective dose and "α" is the slope parameter for Beta-Poisson model. "m" is the slope and "c" is the intercept for linear model, equivalent to 1986-criteria model. "a" is the background probability of HCGI illness in the study population.

The epidemiological study commissioned by USEPA in 1982, recruited a total of 34,598 participants from beaches at Lake Erie, Pennsylvania, and Keystone Lake, Oklahoma. During the study, it was found that sites had a very small count of non swimmers as compared to swimmers. As a result, the non swimming control groups from each beach within a single swimming season were pooled to form a single control population. It was considered that pooling of non swimming control groups for each year increased the probability of detecting a difference in the incidence of illness between swimmers and non swimmers if it does exist. Table 3 summarizes the data collected during that study.

Table 3-3 Summary of 1984-epidemiological study conducted by USEPA

Lake-Beach-Year	**Dose Ent CFU/100ml**	Total Swimmers (S)	Total Non-Swimmers (NS)	Pooled non swimmers $Pool_{NS}$	Illness rate for swimmers per 1000	Illness rate for non swimmers per 1000	**Delta[a] Illness rate per 1000**
Erie A 1979	**5.2**	3020	1310	2349	17.2	14.9	**2.3**
Erie B 1979	**13**	2056	1039	2349	19.5	14.9	**4.6**
Erie A 1980	**25**	2907	1436	2994	16.5	11.7	**4.8**
Erie B 1980	**71**	2427	1558	3086	26.4	11.7	**14.7**
Erie B 1982	**20**	4374	1650	1650	24.9	13.9	**11**
Keystone W 1979	**38.3**	3059	551	970	20.6	15.5	**5.1**
Keystone E 1979	**6.8**	2440	419	970	16	15.5	**0.5**
Keystone W 1980	**23**	5121	774	1211	13.5	8.3	**5.2**
Keystone E 1980	**20**	3562	437	1211	11.2	8.3	**2.9**

The bold entries are the data on which regression analysis was performed and the 1986- Dose Response (DR) model was derived. Two statistics:- correlation coefficient, and the standard error of the estimates, were reported to justify the proposed model (Cabelli, Dufour et al. 1982). Using the above information at each dose level, positive and negative responses were calculated and the model fitting was performed after adjusting for illnesses in the control group. The deviance of the estimated values from the observed values was compared with the critical value of the χ 2 distribution at degrees of freedom equal to the number of doses minus the number of optimized parameters for exponential, beta poisson and linear models (eq-2,3,and 4). The correlation coefficients of the predicted illness rates with respect to the observed illness rates, for each of the three new models (linear, exponential and beta poisson) were compared with the correlation coefficient obtained for the estimates from 1986 linear model.

3.2.4 Probabilistic Analysis

Stochastic models for representing variability and uncertainty in the input data were used to develop GI risk estimates. Under the probabilistic approach, instead of point estimates, input distributions are generated to take the uncertainty and variability into consideration for each input parameter. Thus, probability distributions were developed for each observed parameter (users proportional distribution and exposure duration distribution) using the distribution fitting feature in Crystal Ball© Pro software. Since fitting in Crystal Ball cannot be applied to data sets with less than 15 points. Hence the water quality data on *enterococci* concentration was modeled using R statistical software instead of using Crystal Ball.

Two risk estimates; total illnesses (illnesses/day) and illnesses per 1000 users (illnesses/1000 users/day), were generated for both dry and wet weather conditions. The total illnesses were determined based on the individual risk estimates. The individual risk estimates for each activity (the per-day, per person risk of illness due to a particular recreational use of water) were first defined by a Monte Carlo (MC) analysis in which dose was calculated by randomly drawing values from the concentration distribution, ingestion rate distribution and exposure duration distribution. These values were substituted into the appropriate dose response models to predict the probability of a person getting ill due to a particular water recreational activity in the study water. The observed risk of illness at a particular site due to a certain activity was calculated by drawing values from the use frequency distribution for that activity at that site. The results for all the observed activities at a site were combined to predict the total illness at that site.

The second set of analyses to predict expected number of illness for 1000 users was conducted in similar fashion. The results of the individual risk estimates per activity, as calculated earlier were now multiplied by 1000 to estimate risk of illness per activity per 1000 users. The surveillance information was used to estimate the proportion of users participating in each observed activity at each site. The risk of illness per activity per 1000 users for all the observed activities at a site was combined by setting up a uniform distribution between 0 and 1 from which values were randomly chosen. To this uniform distribution, the proportion of users participating in a particular activity was applied so that if the chosen value falls within the range of the user's proportion for a particular activity, then they were considered to have engaged in that activity, and incurred the concomitant risk. All the simulations were done by drawing 10,000 samples for the analysis. The flow chart representing the calculation approach is shown in figure 3.1.

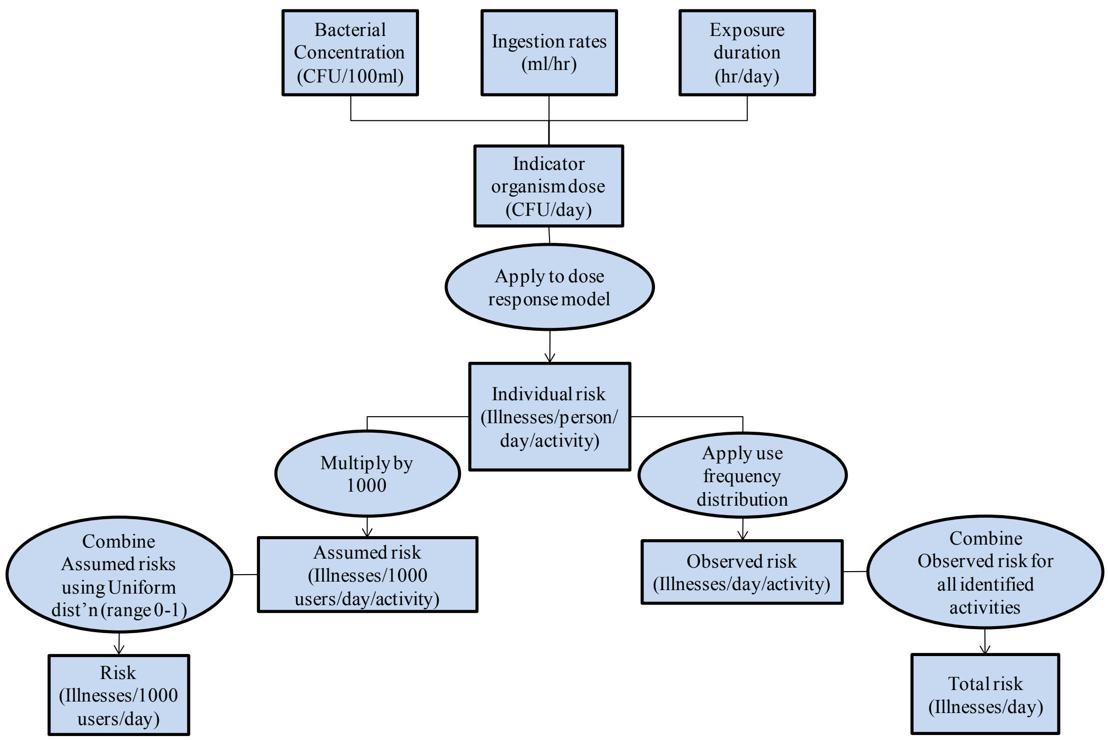

Figure 3-1 Flow chart for calculating total risk of illnesses per day and illnesses per 1000 users per day at each site.

3.3 RESULTS AND DISCUSSION

3.3.1 Exposure Assessment

The water quality information as provided by Philadelphia Water Department for each study water was used to generate bacterial concentration distributions. Three statistical distributions;

Weibull, Gamma and Log normal were fit to the data using maximum likelihood estimation in "R". The fits of the three candidate models were compared and the model providing the best fit was assumed to characterize the indicator organism distribution at the site best. The goodness of fit for all three distributions was based on the Kolmogorov-Smirnov (K-S) test. From this test, the distribution which returned the lowest K-S test statistic was assumed to provide the best fit. For all the sites under both the weather conditions, the lognormal characterized the bacterial distribution best. Table 3.4 represents the range of enterococci concentrations for dry and wet weather conditions at all the study locations. In each sampling event the volume collected was 100 ml and the detection limit of the membrane filtration method was 1 CFU.

Table 3-4 Summary of Enterococci concentrations in CFU/100ml for dry and wet weather conditions.

Water body type	Sampling Year / Recreational Sites	2007				2010		2011	
		Dry (n[a]=20)		Wet (n=10)		Dry (n= 5)		Wet (n= 6)	
		range	Geo mean	range	Geo mean	range	Geo mean	range	Geo mean
Creeks	AD	40 - 8100	816.1	1140 - 29800	6507.75	NS[b]	NS	NS	NS
	BH	40 - 8900	321.2	340 - 34040	3892.93	NS	NS	NS	NS
	COB	120 - 8400	546.4	440 - 22500	4623.86	NS	NS	NS	NS
Schuylkill River	FD	10 - 1200	40.2	10 - 200	41.47	NS	NS	NS	NS
	BG	5 - 1380	29.4	10 - 1700	217.57	NS	NS	NS	NS
Delaware River	PH	NS	NS	NS	NS	2- 14	5.5	560 - 2020	773
	PP	NS	NS	NS	NS	4 - 24	10.1	1810 - 5550	2478.7
	FA	NS	NS	NS	NS	2 - 8	4.3	708 - 1080	932
	PT	NS	NS	NS	NS	2 - 44	8.7	488 - 980	634.5
	PL	NS	NS	NS	NS	4 - 14	9.2	240 - 1160	533.4

[a] Number of samples collected for the given weather condition during that year
[b] NS stands for No Sampling for that period.
[C] Wet weather 2010 data was excluded from the risk calculation due to very less data points (n=2).

During 2007, the average bacterial concentration in creeks was 16% and 39% greater than the average concentration in the Schuylkill River for dry and wet weather conditions, respectively. This bacterial characteristic can be attributed to the physical property of the waterway system. In general, due to varying loading and dilution ratios the concentration in large water bodies will tend to be lower than the concentrations in small creeks. However, the wet weather geometric means for Delaware River sites were found significantly elevated than the Schuylkill River sites, indicating higher potential of pollution at Delaware River sites under wet-weather discharge conditions. The wet weather concentration at creeks was found significantly greater than dry weather concentration by 9%, with p-value as 0.02. However, this difference was not significant at Schuylkill River sites (p-value as 0.12). Unfortunately, for the Delaware River a strong inference on dry-wet weather comparison cannot be made due to temporal variation in the dataset (dry weather-2010 v/s wet weather-2011). Interestingly, according to EPA advisory limit on freshwater-recreation (single sample maxima 61 CFU/100ml), only the Delaware River sites under dry weather condition meet the criteria based on the data we collected.

Based on the usage data presented earlier, empirical distributions were developed for each activity in Oracle's Crystal Ball© Pro software. The final mean estimates of the user's proportional distribution applied in the risk model are presented in Table 5 below. Two sample student t-tests at 5% significance level were done to confirm if the dataset under dry and wet weather condition from two different sites can be treated as the same. Additionally, the activity duration distributions were developed for both transient and non-transient activities via the KS goodness-of-fit test by using maximum likelihood estimation (Table 3.6). Since the cameras were stationary, the duration of certain mobile transient activities (boating, kayaking, canoeing and jet skiing) were calculated differently than the sum of time recorded on film used for non-transient activities (wading, playing and fishing). Considering the comparatively similar extent of water-exposure in water skiing, wind surfing, tubing and swimming, these were grouped together in one category, "swimming". A two sample student t-test was conducted to determine whether the duration data for a particular activity from each site were different. As mentioned earlier, data from two sites (sites PT and PL) were not available previously while performing the pooling test. Hence, revised duration distributions were obtained after considering the new data from these two sites. The final duration distributions used in our risk calculation are summarized in Table 3.6. For detailed discussion about the user's frequency, duration estimates and the weather influence on usage statistics please refer to our previous publication (Sunger, Teske et al. 2012).

Table 3-5 Average proportion of users engaged in each activity.

Recreational Use activities	FD		BG		PH		PP		FA		PT		PL		COB		AD		BH	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Jet skiing	7.90%		3.20%	0.50%	10.00%	8.00%	2.00%			2.30%	6.20%		2.60%		n/a		n/a		n/a	
Wading	n/a		n/a		n/a		8.00%		n/a		n/a		n/a		56.00%	42.00%	57.00%		94.00%	100.00%
Swimming	n/a		n/a		1.00%	0.49%	2.00%			1.00%	n/a		n/a		n/a		n/a	2.00%	n/a	
Boating	22.30%		28.46%	17.03%	82.00%		52.00%		75.40%	65.40%	67.50%		89.00%		n/a		n/a		n/a	
Fishing	57.30%		67.40%	81.30%	4.00%	6.90%	26.00%		20.50%	30.30%	12.70%		n/a		5.00%	9.00%	20.00%		6.00%	0.00%
Boat Fishing	12.50%		0.84%	0.83%	1.00%		n/a		n/a		2.80%		n/a		n/a		n/a		n/a	
Playing with water	n/a		n/a	0.33%	2.00%		10.00%		1.00%		10.80%		n/a		39.00%	49.00%	23.00%	21.00%	0.00%	0.00%
Duck Tours[a]	n/a		n/a		n/a		n/a		n/a		n/a		8.60%		n/a		n/a		n/a	

[a] For users count per boat, only people sitting in front row (n=4) were considered to get wet while the boat splashes into the river to start the tour (based on interview of "Ride the Duck" touring crew members.

Table 3-6 Exposure duration distribution by activities (hrs/day).

Exposure Durations for observed activities based on camera observations, Modeled Distribution(parameters)[e] (hrs/day)																	
Activities		Site FD[a]		Site BG[a]		Site PH[b]		Site PP[b]		Site FA[b]		PL[b]		PT[c] (mean [95% UCL])		Creeks[d]	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Transient	Jet ski/Kayak	Tri(1,2,6)				LN(0.31,0.52)		Wei (0.06,1.56)		Wei(0.13,1.17)		LN(0.13,0.11)		0.18 [0.80]		N/A	
	Boating	Tri(2,3,6)				Wei(1.35,1.06)	LN(0.8,1.45)	Exp(2.6)		Wei(1.1,1.09)		Beta(0.29, 8.5, 0.74, 4.1)		0.73 [4.75]		N/A	
	Boat Fishing					Max ext (2.26,1.3)		N/A		N/A		N/A		0.13 [0.49]		N/A	
	Duck tours	N/A				N/A		N/A		N/A		Tri (0.13, 0.17, 0.2)[f]		N/A		N/A	
Non-transient	Wading	N/A				N/A		Wei(0.2,0.73)		N/A		N/A		N/A		LN(0.132, 0.145)	
	Fishing	G(1.3, 2.0)				Wei(2.32,1.61)				G(1.09,1.56)		N/A		1.59 [4.52]		G(0.44, 1.16)	
	Playing with water	N/A				LN(0.14,0.31)		Wei(0.31,0.89)		Wei(0.05, 3.01))		N/A		0.17 [0.61]		LN(0.368, 0.642)	
	Tubing/Swimming	N/A				LN(0.07,0.04)		G(0.39,1,12)		Wei(0.05, 1.87))		N/A		N/A		N/A	

[a] Duration of transient activities is based on the professional judgments provided by the Schuylkill River Development Council personnel.

[b] Duration of transient activities represents the person hour of use of a particular activity at a site.

[c] In absence of camera observations at PT, duration statistics represents the average and upper 99.73 percentile values, based on 15 in-person surveys, each of 3 hour duration.

[d] Common duration distributions for each activity at sites along the creeks (AD,BG, COB) were used (p>0.05)

[e] Tri-Triangular(min,mode,max); G-Gamma(scale,shape); LN-Log normal(mean, SD); B-Beta(min,max,alpha,beta); Wei-Weibull(scale,shape); Exp-Exponential(rate); Logistic (mean,scale); Min. &Max. Extreme (likeliest, scale).

[f] Duration distribution for duck tours is based on the phone interview of Ms. Best at "Ride the ducks" touring company (likeliest duration-10 min in water).

3.3.2 Dose Response Assessment

The test statistics obtained to assess an alternative model for representing the relationship between the bacterial data and the probability of GI illnesses are shown in Table 3.7.

Table 3-7 Summary of MLE analysis related to Enterococci concentrations and swimming-associated GI illnesses based on 1986-USEPA epidemiological studies.

Alternative DR Models	Minimized Deviance	Parameter Estimates	Std. Error of estimates	χ^2 Upper 95th percentile
Linear Model	34.15	m= 7.68; c= - 3.96	SEEm= 1.09; SEEc=1.22	14.067
Exponential Model	34.16	a=0.0015; k=0.00018	SEEa=0.5; SEEk= 0.19	14.067
Beta poisson Model	34.05	a=0.0022; α = 0.16; N50=59938	SEEa=0.6; SEEα=0.75; SEEn50=2.49	12.592

A significantly acceptable fit must have a deviance lower than the critical values of the chi-square distribution at 95% confidence level ($\chi^2_{0.95,df}$). Since the optimized deviances are greater than the chi square values, none of the tested model is a good fit to the data, including the linear model which is equivalent to the 1986-criteria model (Table 3.7). However, when we check the Pearson correlation coefficient between the observed probability of illness and the predicted probability of illness from all the three models, it is consistently good with values such as: 0.7992 and 0.7937 for exponential and beta poisson models respectively as compared to the 1986-criteria model (R = 0.7552). Thus, in absence of a best fit dose-response model, all the models are equally acceptable with good correlation coefficients. Therefore, in the present study we selected the exponential model to conduct the risk calculations as Beta Poisson is a more complex model with no statistically significant improvement in the goodness of fit, and the linear model proposes a threshold dosage value which is fundamentally implausible. However, we also performed the calculations using the U.S. EPA log linear model to evaluate the difference in two estimates but the results, calculated only from an exponential model were further examined to define the disease burden from different activities at each study location.

3.3.3 Risk Characterization

Total illnesses and illnesses per 1000 users were determined by Monte Carlo simulation using two dose-response models; Log-linear Model (Eq-1) and Exponential Model (Eq-2). The results summarizing the mean rate of GI illnesses per 1000 users and per observed number of users under dry and wet weather periods are shown in Table 3.8 and Table 3.9, respectively. The bold values represent the site-weather combination when the estimate of illnesses exceeded the acceptable recreational water health risk of 8 GI cases per 1000 users. Since the exponential DR model is considered as more sophisticated model to represent the host-pathogen interaction as opposed to empirical model such as log linear model, we considered this model to be more appropriate in estimating human health risk. Thus, results obtained from the exponential model are discussed in detail in the following section.

Table 3-8 Summary of average GI illnesses per 1000 users, based on user's proportion engaged in each activity during dry and wet weather periods.

Urban Water body	Recreational sites	Average risk of gastroenteritis illnesses/1000 users/day[a]			
		Linear 1986model		Exponential Model	
		Wet[b]	Dry[c]	Wet[b]	Dry[c]
Schuylkill River	Fairmount Dam	3.51	3.77	4.74	**8.62**
	Bartram Garden	6.09	1.62	**24.93**	7.33
Delaware River	Pleasant Hill	5	0.03	6.8	1.95
	Pennypack	8.02	0.04	**17.55**	1.76
	Frankford arsenal	6.42	0.0001	**8.58**	1.72
	Penns Landing	3.85	0.0000	5.28	2.01
	Penn Treaty[d]	1.97	0.07	4.63	2.19
Tacony Creek	Adams	**11.16**	4.16	**49.77**	**11.04**
	Bingham	**8.62**	1.86	**48.29**	4.8
Cobbs Creek	Cobbs	**8.32**	3.25	**51.9**	7.8

[a] Bold values indicate the sites where the risk of GI illnesses was above EPA limit

[b] Wet weather bacterial data for Schuylkill River, Tacony Creek and Cobbs Creek was from 2007 sampling, while for Delaware River sites it is from 2011 sampling.

[c] Dry weather bacterial data for Schuylkill River, Tacony Creek and Cobbs Creek was from 2007 sampling, while for Delaware River sites it is from 2010 sampling

[d]Average dose was calculated instead of a probabilistic dose distribution, due to limited data on exposure duration (only in-person surveys were conducted at this site).

Risk per 1000 users: According to the linear model, the only sites presenting risk greater than the EPA limit were those located along the creeks and it happened during wet weather conditions. However, using the exponential model, sites BG, PP and FA also violated the safe-recreational water-criteria. As an exception, site FD showed greater dry-weather risk than the wet-weather risk, causing this site to exceed the criteria marginally under the dry conditions ($risk_{dry}$ FD as 8.62/1000users/day). This disparity is related to the dry weather enterococci levels which were higher than those measured under wet weather conditions, with a maximum value of 1200 CFU/100ml as compared to 200 CFU/100ml (Table 3.4). On the contrary, the sites along the creeks showed higher levels of bacterial concentrations under wet weather conditions and hence presented significantly elevated risk potential. However, the dry weather bacterial count at these sites was also above the EPA limit but with regards to the risk potential; site COB and site BH were found in compliance with the EPA recreational health standard. Knowing the fact that all the sites at creeks experienced comparatively similar usage patterns, with wading being the prominent activity across all sites (Table 3.5), it is appropriate to consider that water quality difference under dry weather condition might have resulted in the risk disparity at these sites (Table 3.4). The dry weather bacteriological count at site AD being more than twice of that measured at other two sites could have contributed to the high level of projected illnesses at this site. The associated contribution of each recreational activity to the final wet-weather risk estimates based on the exponential model is presented in Fig 3. 1.

Interestingly for the Delaware River, the elevated levels of wet-weather bacterial concentration resulted in elevated risk only at two sites (PP and FA). At these two sites, the major contribution in the risk was found due to on-shore fishing (Fig.3. 1) which was the longest duration activity at these sites. Though the most frequent activity on all sites was boating, but it did not lead to an exposure dose that is high enough to contribute significantly in the total risk. At site PP the second activity that contributed significantly to the risk was swimming/tubing. This is obviously related to the higher ingestion volume for direct contact activities (such as swimming and tubing) compared to all the other recreational activities observed at these sites. On the other hand, a lower proportion of users was observed fishing at the other three sites (site: PH, PT and PL) than those observed at site PP and FA (Table 3.5). This coupled effect of fewer people engaging in longer duration activities could have resulted in lower risk potential at these three sites. The wet-weather activities that contributed the maximum risk of GI illness at creeks were wading and playing in water (Fig.3. 1). Similarly, fishing was found to be the potential activity that added the greatest probability of GI illness in the final risk estimate for Schuylkill River sites (Fig. 3.1).

Total illnesses per observed users: In order to understand the health hazard to the users under the current scenario, it was deemed appropriate to have estimates of expected illnesses at each site using observed users frequency. The predicted mean expected cases of GI illness per observed number of users ranged from 0.00 to 1.43 during wet weather and from 0.05 to 0.43 during dry weather (Table 3.9). As observed earlier, risks predicted using the exponential model was higher than those obtained using linear model. As a conservative approach with respect to health protection of users, we chose to also present the 95% confidence interval (CI) range of expected illnesses at each site using the more stringent model (exponential model) for worst weather condition, i.e. for wet weather periods (Fig 3.2). The extent of the difference in the spread of the estimated risk at a water body varied between sites, but the mean value of total expected risk was typically low. The levels of risk posed by users at creeks were comparatively higher with great variability (Fig. 3.2). The highest level of risk estimate was found at site AD, with 95% UCL reaching almost 5 illnesses per day. It is important to note that swimming was also observed at this site, which was not present at other creek-sites (Table 3.5, Fig. 3.1).

Table 3-9 Summary of Total GI illnesses per day based on observed count of users during dry and wet weather periods.

Urban Water body	Average risk of gastroenteritis illnesses/obs users/day[a]				
	Recreational sites	Linear 1986model		Exponential Model	
		Wet[b]	Dry[c]	Wet[b]	Dry[c]
Schuylkill River	Fairmount Dam	0.13	0.09	0.14	0.32
	Bartram Garden	0.14	0.07	0.44	0.12
Delaware River	Pleasant Hill	0.56	0.01	0.54	0.3
	Pennypack	0.47	0.00	0.98	0.1
	Frankford arsenal	0.82	0.00	1.05	0.24
	Penns Landing	0.77	0.00	1.11	0.43
	Penn Treaty	0.17	0.01	0.21	0.1
Tacony Creek	Adams	0.19	0.05	1.43	0.13
	Bingham	0.01	0.02	0.05	0.05
Cobbs Creek	Cobbs	0.09	0.03	0.59	0.08

[a] Total expected illnesses, considering the observed count of users at each site.

[b] Wet weather bacterial data for Schuylkill River, Tacony Creek and Cobbs Creek was from 2007 sampling, while for Delaware River sites it is from 2011 sampling.

[c] Dry weather bacterial data for Schuylkill River, Tacony Creek and Cobbs Creek was from 2007 sampling, while for Delaware River sites it is from 2010 sampling

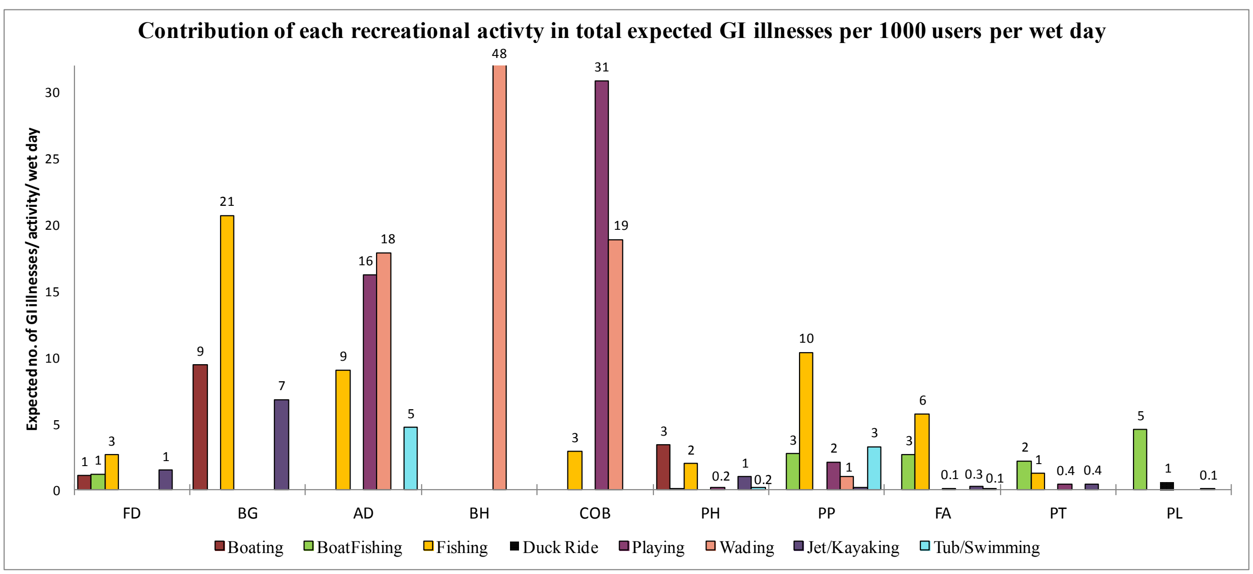

Figure 3-2: Contribution of each recreational activity to the probability of GI illness per 1000 users under wet-weather condition, using exponential dose response model.

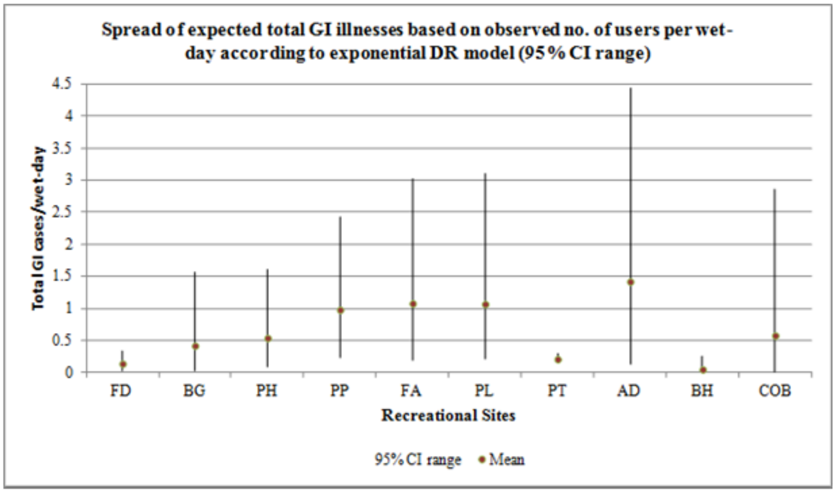

Figure 3-3: Estimated range of GI cases (mean and 95% confidence interval) for observed number of users, including the proportion of frequency of use for each activity, under wet-weather conditions, using exponential dose response model and *enterococci* as fecal indicator.

3.3.4 Uncertainty and Sensitivity Analysis

A sensitivity analysis was performed for all the sites under wet weather conditions for the exponential dose response model. The sensitivity was calculated by computing rank correlation coefficients between every input distribution and the final risk outcome while the MC simulation was running in Crystal Ball © Pro software. Through evaluating the correlation coefficient, we can have a better understanding of the extent by which the input parameters and the risk estimates can change together. The particular input element that has the highest correlation coefficient will have the significant impact on the risk estimate due to both the associated uncertainty in estimating that input parameter and its model sensitivity. The amount of contribution of the three most important uncertain inputs to the variance of the resulting risk of illnesses per 1000 users is shown in Table 3.10.

The model input distribution that contributed the greatest to the variance differs depending on the input source and the type of water body. In general, for sites along the creeks and Schuylkill River the bacterial density distribution contributed the most to the total uncertainty of the risk estimates (average across all sites: 55%, range: 36% to 79%). The user's-proportional distribution contributed on average 18% (6% to 34%) and the exposure duration distribution contributed around 12% (8% to 16%). However, for a large water body, such as the Delaware River, both the water quality parameter and the exposure factors appeared equally potential at influencing the risk distribution.

Table 3-10 Summary of top three input parameters with highest influence on per 1000 risk estimates.

Waterbody	Site	Top three uncertain input parameters for risk estimates	Contribution ToVariance	Rank Correlation
Schuylkill River	FD	Ent. Conc (CFU/100ml)	44%	0.587
		Fish proportion	27%	0.462
		Boating Proportion	9%	0.266
	BG	Ent. Conc (CFU/100ml)	79%	0.841
		Fishing Duration (hrs/day)	8%	0.270
		Fish proportion	6%	0.227
Delaware River	PH	Ent. Conc (CFU/100ml)	29%	0.469
		Boating Duration (hrs/day)	27%	0.450
		Fish proportion	20%	0.392
	PP	Fish proportion	35%	0.531
		Ent. Conc (CFU/100ml)	22%	0.425
		Fishing Duration (hrs/day)	18%	0.387
	FA	Fish proportion	40%	0.589
		Fishing Duration (hrs/day)	28%	0.491
		Boating Duration (hrs/day)	11%	0.310
	PL	Boating Duration (hrs/day)	60%	0.750
		Ent. Conc (CFU/100ml)	28%	0.513
		IR boating (ml/hr)	5%	0.214
	PT	Boating Proportion	29%	0.511
		Ent. Conc (CFU/100ml)	27%	0.487
		Fish proportion	22%	0.439
Creeks	AD	Ent. Conc (CFU/100ml)	36%	0.535
		Playing Proportion	19%	0.388
		Wading proportion	16%	0.357
	BG	Ent. Conc (CFU/100ml)	79%	0.874
		Wading duration	16%	0.392
		IR Wading	5%	0.208
	COB	Ent. Conc (CFU/100ml)	37%	0.536
		Playing Proportion	34%	0.513
		Wading proportion	19%	0.382

The sensitivity results from the Delaware River indicate almost 31% contribution from the users proportion distribution (range: 20% to 40%), while 29% from the duration distribution, followed by 27% contribution from the bacterial density distribution. In order to account for the water quality parameter, we increased the number of water samples at a few sites and performed preliminary risk estimates during the study period. But with an increased number of water samples, the extent of the uncertainty did not decrease from the previous results (data not shown). Thus, it can be interpreted that there is intrinsic variability in the water quality data which cannot be controlled from getting propagated into stochastic models developed for environmental risk assessment.

Unlike variability, which is a natural variation in the phenomenon of interest, uncertainty represents our lack of knowledge about the phenomenon (Hamilton, Stagnitti et al. 2006). We collected recreational use information using time-lapse cameras which provided comprehensive exposure statistics for non-swimming recreational exposures. However, the limitations associated with relying on remote surveillance; such as the camera's restricted field of view and loss of data due to time gaps between film shots resulted in large uncertainties in estimating the exposure durations. This influence was dominant at sites where field data was collected to estimate the person-hour of use of a particular transient activity, such as sites along the Delaware River. A greater proportion of transient activities was observed at these sites as compared to sites along other water bodies. The duration distribution (especially boating distributions) contributed maximally to the variance in risk estimates for the Delaware River sites. We have previously discussed in detail the sources of uncertainty in collecting this exposure information (Sunger, Teske et al. 2012). Briefly, for calculating transient activity duration, a different duration definition was used which represented the person hour of use at these sites. This approach represented the sum of time frames for which a particular activity was seen at a site, instead of calculating the actual length of time for which one particular person will be engaged in a specific activity. On the other hand, transient activities were also observed at Schuylkill River sites but this river is comparatively smaller than the Delaware River and the river mile under investigation had just one boat/jet ski dock. With a single point of entrance to the river stretch, it was possible to get a fairly good estimate for the total time of boat/jet ski tours through expert interviews.

The other input parameter causing considerable variance in risk outputs was the user's proportion distribution. This information was also collected through camera surveillance and was validated with in-person surveys. Although a wide range of observations were collected over consecutive years to define usage frequency at these sites, it is still is conceivable that a certain proportion of the total uncertainty associated with user's frequency data represents true uncertainty. However, unlike the duration distribution uncertainties, it is probably reasonable to assume that a part of the usage frequency uncertainty could be attributed to human variability in preference to a recreational activity under varying environmental factors, such as- high tide v/s low tide or windy v/s calm periods or high v/s low barometric pressure etc.

3.4 CONCLUSIONS

The influence of bacterial load and the usage profile on risk estimates was thoroughly explained in the study by performing probabilistic risk assessment. In this study we assessed the magnitude and probability of potential hazard to public health posed by recreation in surface water bodies impacted by wet weather discharges. Higher risks were found using the exponential model than using the 1986-log-linear criteria model. Exponential model was considered as an alternative to the criteria model due to its acknowledged applicability for modeling health risks and because it is a non-threshold model, which implies that there is no minimum infectious dose.

Our analysis indicated that the risk of GI illness associated with human exposure to Philadelphia's recreational waters, which are impacted by wet weather discharges are of immediate concern. Based upon the per 1000 health risk estimates, the existing water environment at certain locations was incompatible with the federal safe-recreational water objectives. This analysis clearly demonstrated that the major creeks in the city, which are very easily accessible for direct water-contact, had episodic periods of poor water quality which presented significantly high level of risk to its users. Though the usage pattern was predominantly "non-swimming", still the concentration of bacteria coupled with recreational-usage demand was found capable of causing gastrointestinal illness. Activities contributing maximum to the risk of GI illness were identified as wading and playing with water, for all the sites along the creeks. The locations monitored were within 200 feet of an existing CSO and surprisingly had high concentrations of fecal indicator organism (enterococci) under both the weather conditions: dry and wet weather. However, significantly low dry-weather risk estimates highlight the fact that relying entirely on the bacterial count for evaluating the health hazard of a water body may sometimes mislead as the administered ingested dose would vary with respect to the type of the recreational usage at that water body. Similar interpretation was made by analyzing the results obtained from risk assessment of large rivers. Though the wet weather samples at both the rivers exceeded the EPA standard for safe recreational water, the elevated risk of GI illness was observed only at the sites where large proportion of users were engaged in on-shore fishing for long durations; ranging from 1 hour to almost 5 hours a day.

The actual cases of GI are dependent on real count of users that are exposed to the water body. Thus, the second set of analysis reported the expected probability of illnesses considering the observed number of users at each site. Though the average risk estimate at most of the sites was less than 1 illness per day, at some sites the 95% maxima reached to 5 illnesses a day. It should be noted that there were potential sources of uncertainty in the risk model used to predict the GI illnesses at study locations. The parameters causing largest uncertainty in the risk estimates were: 1) microbial water quality data and, 2) the exposure duration of transient activities. Though under the natural environmental settings it was out of scope to control the variability associated with the bacteriological data, more information on the transient activity duration may further refine the results significantly. There were multiple boat launch sites at the Delaware River, leading to mixed water traffic in the river. As a result, with camera surveillance it was difficult to calculate the actual per person duration of transient activity in this river. Different surveillance techniques such as: direct interviews with the boaters and managers of launch docks may be applied in future for increasing the confidence in the model output. Furthermore, the geographical feature such as presence of beach area along the water edge was found to significantly increase the opportunity for direct-water contact activities such as: playing and swimming, at large river bodies, thereby presenting greater risk of GI illness to the users.

While interpreting the results obtained by this QMRA study it should be noted that many conservative assumptions were made during the analysis. Due to relatively dry summers during the study period, precipitation of at least 0.1” was considered a wet weather event and any exposure to a water body during and 72 hours after the onset of rain was considered as wet weather exposure. Additionally, the water sample resulting in undetectable bacterial counts was reported as having 10CFU/100ml (half of the detection limit). Factors such as immunity, bacterial re-suspension and incubation period were not included in the model. Such assumptions may overestimate the maximum extent of exposure and thus the associated risk. However, assuming ingestion as the only route of exposure may have underestimated the risk as hand-to-mouth transfer of pathogens and consumption of fish could also lead to increased risk of illness.

The QMRA model developed in this study is beneficial for its ability to incorporate variability from all input parameters, including swimming and non-swimming exposure scenarios and to project the total health risk at a water body under different weather conditions. This model provided a useful scientific tool for managing risks associated with the non-swimming recreational uses of water bodies. As a result of this analysis numerical estimates of health risks were obtained and the health risk associated with potential exposure to site-related usage pattern was characterized. The information about the influence of usage statistics on the risk potential at a particular water body can be useful for policy makers in planning the health campaigns and prioritizing the future interventions. The QMRA model presented in this study can be further refined to accommodate potential pathogens present in the recreational waters. Researchers have investigated likely concentrations of other primary etiologic agents in recreational fresh waters which would result in the same illnesses as observed in early epidemiological studies (Soller, Bartrand et al. 2010). Many studies have characterized the recreational risk with respect to potential pathogens present in the water body (Donovan, Unice et al. 2008; Rijal, Tolson et al. 2011). Thus a multi-stage QMRA approach, following the same framework as discussed here, may also be applied to Philadelphia waters if comprehensive water quality data is collected in future.

4 Chapter 4: Quantitative Microbial Risk Assessment for recreational exposure to water bodies in Philadelphia, based on *E. coli* model

4.1 Introduction

The epidemiological studies of the1980's recommended the use of *E. coli* and Enterococcus as the fecal indicator organisms to assess the health risks associated with recreation in sewage-contaminated surface waters. The quantitative analysis of the data collected during those studies represented a positive correlation between the instances of swimming-associated gastroenteritis and enterococci/*E. coli* densities; making these organisms a better indicator of pathogen load than fecal coliforms (U.S. EPA 1986). However, there is a large body of scientists who are still unconvinced on the relevance of these FIB's in monitoring the water quality due to the risk of pathogenic viruses and adverse environmental conditions (Byappanahalli and Fujioka 1998; Desmarais, Solo-Gabriele et al. 2002; Yamahara, Sassoubre et al. 2012). Like pathogens, these indicator organisms are present in feces of humans and other warm-blooded animals. However since sampling and culturing all potential pathogenic organisms is uneconomical and non-feasible, indicator organisms are used to assess microbial exposure.

Based on the recent analysis done by US EPA for updating the 20+ years old water quality monitoring guidelines, *E. coli* continues to be the best indicator of fecal contamination in fresh waters while enterococci are recommended as best for saline waters, with a significant scope of application in fresh water as well (U.S. EPA 2011). States are still allowed to use their old criteria for water quality monitoring which are generally based on total or fecal coliform counts. However, it is imperative that they meet the goals of Clean Water Act (CWA) and confirm that the recreational waters are "swimmable" and "fishable". As stated earlier, the state of

Pennsylvania falls under this category; it provides general criteria for bacterial concentrations in recreational waters which rely upon fecal coliforms as indicator organisms (Pennsylvania Code 1971). Under situations where a quantitative estimate of health risk hazard from recreational water contact is desired, the US EPA recommends that the State bacteriological monitoring should switch to *E. coli* or enterococci. This transition is required since to derive the health risk estimate one needs to apply QMRA (Quantitative Microbial Risk Assessment) approach, which uses a probabilistic framework to predict the range of expected illnesses by using the appropriate dose-response models that relates the ingested dose of organism to the adverse health outcomes. Since, the two FIB's have established dose-response models their use as bacteriological monitoring unit would allow states to quantify the risks for water-contact sports in sewage-impacted water bodies.

In the previous chapter we did the analysis based on enterococci data and presented the results using enterococci dose-response model. We further wanted to evaluate the risk potential of the study locations using *E. coli* concentrations. Both the indicator organisms have been studied extensively for their survival and re-growth kinetics in the environment and have raised lots of concern about their applicability in representing true water quality conditions. A study done in Lake Michigan showed that enterococci (ENT) can survive longer than *E. coli* (EC) in freshwaters and that the sunlight intensity plays very important role in inactivation of these indicator organisms (Kenny, Barber et al. 2009). Many other researchers have reported that EC is inactivated more rapidly in marine waters than in freshwaters (MacKichan 1961; Kenny, Barber et al. 2009). Some studies have also reported that under more turbid conditions, EC survives longer than ENT and can result in altering EC to ENT ratio in surrounding environment (U.S. EPA 1972; Ausubel and Grubler 1995; Roberts Kenneth 2006). Due to ongoing uncertainty about the choice of appropriate fecal indicator organism and lack of developed dose-response models for other potential indicator organisms, we investigated both - EC and ENT, to get better understanding about the hygienic quality of Philadelphia waters. Thus in this chapter, the risk analysis was performed on EC data using EC dose response models and then the estimates were compared with those obtained from ENT data.

4.1.1. Indicator Organisms

E. coli is a single species of fecal coliform bacteria while enterococci represent a subgroup of the fecal streptococcus group. *E. coli* and ENT are universally accepted as indicators of fecal contamination in both drinking and recreational waters. They closely meet the indicator organism criteria which were outlined by WHO (2003) for drinking water protection and also listed by Cabelli (1977) for monitoring recreational waters. Though for drinking water, total coliforms are still the standard test because their presence indicates contamination of a water supply from an outside source. The requirements for an indicator as listed by Cabelli are as follows (Cabelli 1977) :-

A. The best indicator organism should be the one whose densities correlate best with health hazards associated with one or several given types of pollution sources.
B. The indicator should be consistently and exclusively associated with the source of the pathogens.
C. It must be present in sufficient numbers to provide an accurate density estimate whenever the level of each of the pathogens is such that the risk of illness is unacceptable.

D. It should approach the resistance to disinfectants and environmental stress, including toxic materials deposited therein, of the most resistant pathogen potentially present at significant levels in the sources.
E. It should be quantifiable in recreational waters by reasonably facile and inexpensive methods and with considerable accuracy, precision, and specificity.

In addition to these guidelines it is also recommended by WHO that the indicator should not be a pathogenic micro-organism (to minimize the health risk to analysts) and it should not multiply in the environment (WHO 2003). Although these criteria proved successful in reducing the spread of waterborne diseases, new challenges to public health safety have been identified due to viral and protozoan contamination of recreational waters originating from pollution with human excreta. Moreover, evidence from a number of other studies has suggested that *E. coli* and enterococci may persist and grow in warm, subtropical waters, raising concern about their use as indicator organism. There is a wide range of scientists who argue on extended persistence of culturable bacteria in the sediments of recreational waters and argue on their compatibility to be used as indicators of fecal contamination (Smith 1990; Kramek and Loh 2007; Levine 2010).

4.2 Data Acquisition

The basic framework for the risk assessment is the same as discussed in Chapter -3 (Figure 3.1, Chapter 3). Since the reference organism is changed from ENT to EC; the dose-response model and the water quality information was modified accordingly. The procedure of which is discussed in detail in the following section. All other input parameters to the exposure model such as: users frequency distribution, ingestion rate estimates and exposure duration for each activity, are not subject to change as they are independent of the choice of the indicator organism.

4.2.1. Exposure and Water Quality data

The Schuylkill river and creeks sites (Fairmount Dam (FD), Bartram's Garden (BG) & Adams Avenue dam (AD), Bingham Street (BH), Cobbs educational center (COB)) were sampled during the recreational season of 2007 for bacterial enumeration by Drexel team, while the Delaware River sites (Pleasant Hill Park (PH), Penny pack park (PP), Frankford Arsenal (FA), Penn Treaty Park (PT), and Penn's Landing (PL)) were analyzed in 2010 by Bureau of Laboratory Services (BLS), a state-of-the-art laboratory performing the services for the Philadelphia Water Department (PWD). Due to relatively drier summer in 2010, wet weather microbial data for the Delaware River sites were in-sufficient to conduct probabilistic exposure analysis; hence point estimate of *E. coli* densities were used to develop exposure statistics. Both the sampling events of 2007 and 2010 strictly followed the US EPA recommended microbiology methods manual (Verbrugge, Rainey et al. 2004) and used the EC test-method 1604 for bacteria quantification in collected samples.

The measured bacterial concentrations were divided into dry and wet weather datasets, depending on when the water samples were procured. A wet weather water sample was considered any sample collected during or 72 hours after a rainfall event greater than 0.1", as reported by the Philadelphia International Airport. A lognormal distribution was obtained as a best fit model based on Kolmogorov-Smirnov (K-S) test to describe all the datasets under both dry and wet weather conditions. Table 4.1 represents the distribution functions derived for each site to represent the *E. coli* density variation during the study period.

The majority of all observed recreational activities were categorized as "secondary-contact activities." The user's count, proportional distribution of user's frequency by activity and exposure duration statistics at each site were obtained from in-person surveys and time-lapse camera surveillance. The survey results for all sites, the ingestion and exposure duration estimates for different types of water-based recreational activities have been discussed previously (Sunger, Teske et al. 2012). These datasets were directly applied as the input parameter to the exposure model in order to predict the range of the ingested dose volume.

Table 4-1 Summary of E. coli concentration distributions for dry and wet weather conditions.

E. Coli. density Data			
Water Body (Sampling Year)	Site	Fitted Distribution (Mean, Std. deviation)	
		Dry Weather	Wet Weather
Creeks (2007)	Adams	LN(2.93,0.39)	LN(3.67,0.44)
	Bingham	LN(2.32,0.24)	LN(3.4,0.63)
	Cobbs	LN(2.46,0.49)	LN(3.59,0.56)
Schuylkill River (2007)	Fairmount Dam	LN(1.52,0.44)	LN(2.14,0.43)
	Bartram Garden	LN(1.47,0.5)	LN(2.95,0.29)
Delaware River (2010)[a]	Pleasant Hill	LN(1.3,0.25)	1.40
	Penny pack	LN(1.36,0.28)	1.48
	Frankford	LN(1.27,0.25)	2.00
	Penn Treaty	LN(1.25,0.36)	1.81
	Penns Landing	LN(1.7,0.09)	1.90

[a] Due to limited Wet weather data at Delaware sites, log transformed point estimates were used to represent the E. Coli bacterial quality.

4.2.2 Dose Response estimates

The existing health risk guidelines for both fresh and marine recreational waters are based on dose-response relationship derived from the Dufour 1982 studies. Using regression analysis on the collected data, the researchers proposed a log linear model (equation 1) representing the relationship between concentrations of *E. coli*. or *Enterococci* in the water and the additional risk of gastrointestinal illness in swimmers (Cabelli, Dufour et al. 1982). This model offers a threshold *E. coli* concentration (approximately 17.75 CFU/100 mL) below which no excess risk of HCGI illnesses per 1000 users above the background is projected.

Similar to the analysis done on the enterococci data of 1982 epidemiological studies (Chapter 3), the *E. coli* concentrations were re-analyzed for investigating alternative non-threshold models; including beta-Poisson model (equation 2) and exponential model (equation 3). The method of maximum likelihood estimation and the chi-square goodness of fit test were used following the approach described for conducting a QMRA study (Mahoney, Farley et al. 1992). The results were compared with the 1986-criteria linear model (equation 4). Similar to ENT analysis, we assume an underlying baseline response that is dose-independent, upon which the dose-dependent response is superimposed. All the statistical analysis was done using "R", a statistical program and applying the Nelder–Mead algorithm. Both the candidate distributions assume that the organisms are Poisson distributed in the source-samples and that the probability of illness is binomial/beta distributed (Mahoney, Farley et al. 1992; Teunis, Nagelkerke et al. 1999).

$$\mathrm{P(d)} = \frac{9.4.log10(d) - 11.74}{1000} \qquad \text{Eq (1)}$$

$$\mathrm{P(d)} = a + (1-a).\left(1 - e^{-k.d}\right) \qquad \text{Eq (2)}$$

$$\mathrm{P(d)} = a + (1-a).\left[1 - \left\{1 + \left(\frac{d}{N_{50}}\right).\left(2^{1/\alpha} - 1\right)\right\}^{-\alpha}\right] \qquad \text{Eq (3)}$$

$$\mathrm{P(d)} = m.log10(d) + c \qquad \text{Eq(4)}$$

where P(d) is the probability of response at dose "d" and "k" is the probability that a single organism can survive and initiate infection. "N50" is the median infective dose and "α" is the slope parameter for Beta-Poisson model. "m" is the slope and "c" is the intercept for linear model, equivalent to 1986-criteria model. "a" is the background probability of HCGI illness in the study population. Table 4.2 summarizes the data obtained from the 1982 epidemiological study (Cabelli, Dufour, Mccabe and Levin, 1982) in order to develop revised dose-response model.

Table 4-2 Summary of 1982-epidemiological study conducted by USEPA

Dose **EC CFU/100ml**	Total Swimmers (S)	Total Non-Swimmers (NS)	Pooled non swimmers $Pool_{NS}$	Illness rate for swimmers per 1000	Illness rate for non swimmers per 1000	**Delta [a] Illness rate per 1000**
23	3020	1310	2349	17.2	14.9	**2.3**
47	2056	1039	2349	19.5	14.9	**4.6**
137	2907	1436	2994	16.5	11.7	**4.8**
236	2427	1558	3086	26.4	11.7	**14.7**
146	4374	1650	1650	24.9	13.9	**11**
138	3059	551	970	20.6	15.5	**5.1**
19	2440	419	970	16	15.5	**0.5**
52	5121	774	1211	13.5	8.3	**5.2**
71	3562	437	1211	11.2	8.3	**2.9**

[a] Delta illness rate = illness rate for swimmers- illness rate for non-swimmers

The bold entries are the data on which regression analysis was performed and the 1986- Dose Response (DR) model was derived (Cabelli, Dufour et al. 1982). Using the above information at each dose level, exponential and beta-Poisson models were tested by comparing the deviances of the estimated values from the observed values. The critical value of the χ 2 distribution at degrees of freedom equal to the number of doses minus the number of optimized parameters for exponential, beta-Poisson and linear models was used to test the goodness of fit. The results of the test are shown in table 4.3 and the three model fits with the observed data points are shown in Figure 4.1. The optimized deviances for all the tested models were found greater than the chi square values (Table 4.3). Hence none of the tested models are a good fit to the data, including the linear model, which is equivalent to the 1986-criteria model.

Table 4-3 Summary of MLE analysis related to E. coli. concentrations and swimming-associated GI illnesses based on 1986-USEPA epidemiological studies.

Alternative DR Models	Minimized Deviance	Parameter Estimates	Std. Error of estimates	χ^2 Upper 95th percentile
Linear Model	21.56	m= 8.27; c= - 9.53	SEEm= 1.04; SEEc=1.71	14.067
Exponential Model	19.46	a=0.000105; k=0.0000511	SEEa=0.61; SEEk= 0.15	14.067
Beta poisson Model	19.46	a=0.0000511; α = 1917; N50=13567	SEEa=0.6; SEEα=0.75; SEEn50=2.49	12.592

For further analysis, we compared the correlation coefficients of the predicted illness rates with respect to the observed illness rates, for each of the three new models (linear, exponential and beta Poisson). The results are shown in Table 4.4.

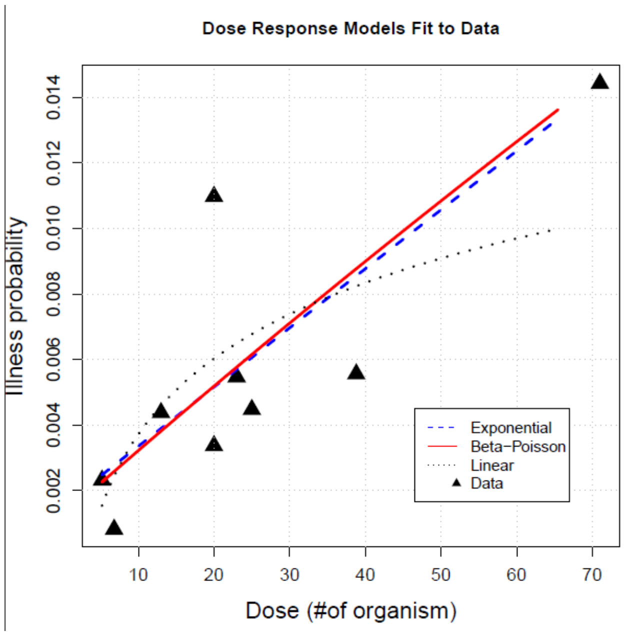

Figure 4-1 Dose Response data and model fits for Highly Credible Gastrointestinal Illness (HCGI) in fresh water swimmers.

Based on the Pearson correlation coefficient between the observed probability of illness and the predicted probability of illness for all the three models, comparatively better results were obtained for exponential/beta-Poisson model than 1986-criteria model (R = 0.8789 for exponential model v/s R= 0.8022 for 1986 criteria model; Table 4.4). Thus, in absence of a best fit dose-response model, all the models are equally acceptable with good correlation coefficients. Therefore, in the present study we selected the exponential model to conduct the risk calculations as Beta Poisson is a more complex model with no statistically significant improvement in the goodness of fit, and the linear model proposes a threshold dosage value which is fundamentally implausible. However, we also performed the calculations using the U.S. EPA log linear model to evaluate the difference in two estimates.

Table 4-4 Summary of comparison between predicted illness rates and observed illness rates for candidate dose-response models.

EC Conc Dose (CFU/100ml)	Response rate per person	Illness rate based on Dufour- linear model	Illness rate based on Haas- linear model	Illness rate based on Haas- Exp. Model	Illness rate based on Haas- BP Model	SSE (obs-pred)^2			
	obs_prob	pred_linear1986	pred_linearNew	pred_exp	pred_bp	SSEln_1986	SSEln_new	SSEexp	SSEbp
23	0.0023	0.0065	0.0017	0.0013	0.0013	1.76E-05	3.60E-07	1.00E-06	1.00E-06
47	0.0046	0.0094	0.0043	0.0025	0.0025	2.30E-05	9.00E-08	4.41E-06	4.41E-06
137	0.0048	0.0138	0.0081	0.0071	0.0071	8.10E-05	1.09E-05	5.29E-06	5.29E-06
236	0.0147	0.016	0.0101	0.0121	0.0121	1.69E-06	2.12E-05	6.76E-06	6.76E-06
146	0.011	0.014	0.0084	0.0075	0.0075	9.00E-06	6.76E-06	1.23E-05	1.23E-05
138	0.0051	0.0138	0.0082	0.0071	0.0071	7.57E-05	9.61E-06	4.00E-06	4.00E-06
19	0.0005	0.0057	0.001	0.0011	0.0011	2.70E-05	2.50E-07	3.60E-07	3.60E-07
52	0.0052	0.0098	0.0047	0.0028	0.0028	2.12E-05	2.50E-07	5.76E-06	5.76E-06
71	0.003	0.0111	0.0058	0.0037	0.0037	6.56E-05	7.84E-06	4.90E-07	4.90E-07
					Sum of deviances=	**3.22E-04**	**5.72E-05**	**4.03E-05**	**4.03E-05**
					correlation coeff=	**0.8022**	**0.8041**	**0.8789**	**0.8789**

4.3 Risk Prediction

Two types of average risk estimates; illnesses/observed no. of users/day and illnesses/1000 users/day, for each site were calculated for both dry and wet weather conditions. The risk estimates were defined by conducting Monte Carlo (MC) analysis in which 10,000 iterations were carried out by randomly drawing values from respective cumulative distributions of each input parameter and substituting them in the exponential dose response model (Eq-2). Similar approach was followed to get estimates based on 1986- criteria linear model (Eq-1). The results summarizing the mean rate of GI illnesses per 1000 users and per observed number of users under dry and wet weather periods are shown in Table 4.5. The bold values represent the site-weather combination when the estimate of illnesses exceeded the acceptable recreational water health risk of 8 GI cases per 1000 users. For risk per 1000 users, the camera surveillance information was used to estimate the proportion of users participating in each observed activity at each site. These observed proportions were then applied to the assumption of 1000 users at each site to calculate expected number of illnesses. It was hypothesized that if 1000 users were recreating at each of these sites, then they would be distributed in the same proportion as observed in the camera survey.

Table 4-5 Summary of average GI illnesses per 1000 users and per observed number of users, during dry and wet weather periods.

Urban Water body	Sites	Risk estimates based on E. coli density water quality data[a]				
		Average daily risk of gastroenteritis illnesses	Linear 1986model		Exponential Model	
			Wet	Dry	Wet	Dry
Schuylkill River	Fairmount Dam	per 1000 users	3.98	0.63	2.98	1.47
		per obs number of users	0.05	0.01	0.09	0.05
	Bartram Garden	per 1000 users	5.77	0.3	7.09	1.71
		per obs number of users	0.1	0.01	0.12	0.03
Delaware River	Pleasant Hill	per 1000 users	0	0.0002	1.33	1.26
		per obs number of users	0	0.0001	0.12	0.19
	Pennypack	per 1000 users	0.0014	0.01	0.07	0.07
		per obs number of users	0.00005	0.0004	1.13	1.18
	Frankford arsenal	per 1000 users	0.04	0.0001	0.19	0.17
		per obs number of users	0.004	0.00002	1.39	1.2
	Penns Landing	per 1000 users	0	0.00004	1.49	1.45
		per obs number of users	0	0.000004	0.32	0.31
	Penn Treaty[b] (average dose)	per 1000 users	0.0002	0.002	1.34	1.29
		per obs number of users	0.000001	0.0001	0.07	0.06
Tacony Creek	Adams	per 1000 users	4.66	1.28	**10.5**	2.45
		per obs number of users	0.1	0.02	0.37	0.03
	Bingham	per 1000 users	3.03	0.04	7.1	1.21
		per obs number of users	0.001	0.003	0.01	0.01
Cobbs Creek	Cobbs	per 1000 users	4.16	0.63	**11.23**	1.91
		per obs number of users	0.05	0.01	0.12	0.02

[a] Bold values indicate the sites where the risk of GI illnesses was above EPA limit

[b]Average dose was calculated instead of a probabilistic dose distribution, due to limited data on exposure duration (only in-person surveys were conducted at this site).

All the sites showed acceptable level of risk for GI illnesses (less than 8 illnesses/1000 users/day) during dry and wet weather conditions, except Adams and Cobbs creek sites. This increase in the chances of illnesses can be attributed to considerably high levels of *E. coli* densities observed at these sites; with an average concentration of 4320CFU/100ml at river sites as compared to 188 CFU/100ml at creek sites, during wet periods (Table 4.1). Additionally, using the 1986-criteria model the average risk values under both the weather conditions were pretty low, presenting no potential health risk to the recreational users. Overall for all the sites, the exponential model presented greater risk estimates than the 1986-criteria model (Table 4.5). This discrepancy is mainly associated with the threshold nature of the 1986-criteria model which consider zero chances of illness at bacterial concentrations below a cutoff value, resulting in much lower risk estimates as compared to those obtained from the exponential; a non-threshold model.

The two priority sites - Adams and Cobbs, were further investigated to evaluate the proportional contribution of each observed activity and to identify the potential exposure scenarios leading to an elevated risk estimates at these sites. Figure 4.2 and figure 4.3 represents the certainty ranges of number of illnesses from each activity in proportion to the user's frequency at the scale of 1000 users for Adams and Cobbs sites, respectively. Playing, wading, fishing and swimming were the common activities observed at this site. The proportion of swimmers was very low; almost 7% (Figure 1.3, Chapter 2) as compared to other activities, resulting in relatively low risk contribution from this activity. A great proportion of users, almost 41% were found wading (Table 2.2, Chapter 2) in water, making this activity the most common activity at Adams followed by playing and fishing. According to the per activity risk analysis, playing and wading both were equally significant activities at this site; each contributing 4 illnesses to the total mean estimated no. of 10.5 illnesses/day (the 90% upper bound value of total estimated illness at Adams is 41, with the relative proportion of 16 and 13 arising from playing and wading, respectively; Figure 4.2). This finding is consistent with the underlying assumption of the ingestion rates derived from the US EPA exposure factor handbook (US EPA 2011) and human health risk assessment guidelines (US EPA 2000). These policy documents suggested greater ingestion rate for wading (10ml/hr) as compared to fishing (3ml/hr), making it present more hazard for contaminant exposure than fishing. Another factor that would contribute to the ingested dose volume is the exposure duration. On comparing the exposure duration data for these three activities (data shown in chapter 2), it appears that amount of time spent fishing and playing at creeks is almost same (26 to 22 minutes) but on an average waders were found to spend only 7 minutes in water (Sunger, Teske et al. 2012). This finding highlights the risk potential of wading at compromised water bodies because people were engaged in this activity for very small amount of time as compared to other activities but still it posed a significant level of health risk to the users.

At Cobbs Creek, the average number of estimated illness was 11.2 cases per 1000 users (Table 4.5) with a 90% upper bound value of 39 illnesses (Figure 4.3). The relative risk contribution from all the observed activities showed the greatest contribution from playing, followed by wading and with a very small number of cases from fishing. The average GI cases per activity (with upper 90% bound estimates) were: playing- 8 (28), wading-3 (11) and fishing- 0.6 (2) cases per 1000 users/day (figure 4.3). These findings can be very helpful for public health official and policy makers in focusing their watershed management efforts more efficiently and providing better measures for human health protection.

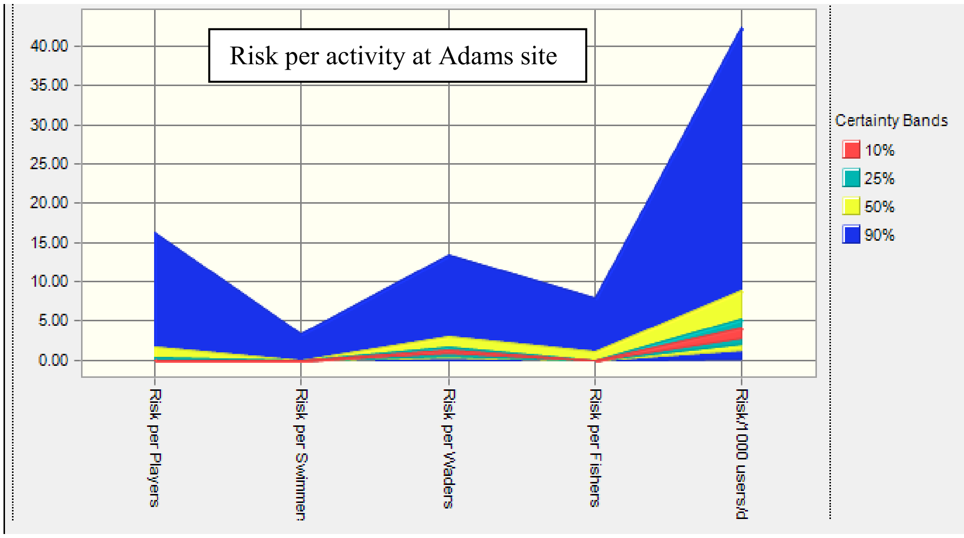

Figure 4-2 Certainty ranges for illnesses predicted by activity at s site for wet weather period per 1000 users per day.

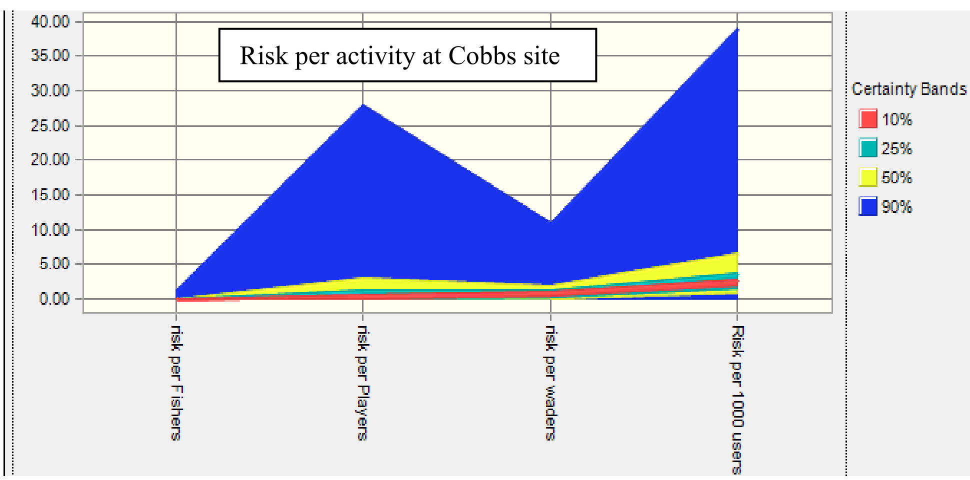

Figure 4-3 Certainty ranges for illnesses predicted by activity at Cobbs site for wet weather period per 1000 users per day.

4.3.1 Uncertainty and Sensitivity Analysis

A sensitivity analysis was performed for all the sites under wet weather conditions using the exponential dose response model. The analysis was conducted by running Monte Carlo (MC) simulations of risk estimates applying input distributions to the exponential dose-response model. The rank correlation coefficients and the variance contribution from each input parameter were obtained as a result of the MC simulations. The amount of contribution of the three most important uncertain inputs to the variance of the resulting risk of illnesses per 1000 users is shown in Table 4.6.

Other than in the Delaware River sites, bacterial density variability contributed most to the model uncertainty (average across all sites: 40%, range: 23% to 66%).; with fishing and wading being the second most important parameters at the Schuylkill River sites and Creek sites, respectively (Table 4.6). Due to point estimates of wet-weather EC densities at the Delaware River sites, the effect of water quality uncertainty could not be evaluated at these locations. However at these sites, the usage frequencies were found very influential in affecting the risk estimates; with the user's proportional distribution (specifically boaters count) being the major uncertainty contributor to risk projections.

Table 4-6 Summary of top three input parameters with highest influence on per 1000 risk estimates, during wet weather conditions.

Waterbody	Site	Top three uncertain input parameters for risk estimates	Contribution ToVariance	Rank Correlation
Schuylkill River	FD	EC. Conc (CFU/100ml)	43%	0.59
		Fish proportion	26%	0.46
		Boating Proportion	8%	0.26
	BG	EC. Conc (CFU/100ml)	42%	0.590
		Fishing Duration (hrs/day)	31%	0.510
		IR Fish (ml/hr)	11%	0.310
Delaware River	PH	Boating Proportion	47%	0.600
		Jet Skiing Proportion	22%	0.410
		Fishing proportion	21%	0.410
	PP	Boating proportion	57%	0.710
		Fish proportion	30%	0.510
		Playing Proportion	7%	0.240
	FA	Fish proportion	54%	0.730
		Boating Proportion	39%	0.610
		Fishing Duration (hrs/day)	3%	0.180
	PL	DuckBoat proportion	64%	0.760
		Boating Proportion	24%	0.470
		Boating Duration (hrs/day)	10%	0.300
	PT	Boating Proportion	29%	0.511
		Playing Proportion	27%	0.487
		Fishing proportion	22%	0.439
Creeks	AD	EC. Conc (CFU/100ml)	24%	0.430
		Wading proportion	23%	0.420
		Playing Proportion	23%	0.420
	BH[a]	EC. Conc (CFU/100ml)	66%	0.800
		Wading duration (hrs/day)	26%	0.500
		IR Wading	8%	0.270
	COB	Playing Proportion	45%	0.590
		EC. Conc (CFU/100ml)	23%	0.420
		Wading proportion	22%	0.410

There is variability associated with bacterial survival in waters, to some extent it can be addressed if large dataset can be obtained both in time and space. For probabilistic estimates of environmental risk modeling, a better understanding of CSO discharges and associated bacterial load variation under different environmental conditions at Philadelphia waters will significantly increase the confidence in our estimates. Furthermore, the uncertainty associated in interpreting camera images for quantifying users count in transient activities (boating, jet skiing etc) also impacted our risk estimates significantly at locations where boating was the most frequent activity; such as Delaware River sites.

4.3.2 E. coli v/s Enterococci risk estimates

Since both the indicator organisms – *E. coli* and enterococci are deemed acceptable indicators for monitoring fresh recreational waters; we compared the results from both the organisms to have an overall estimate about the health risk potential to Philadelphia users. Moreover, the risk estimates during wet periods were of main concern due to high possibility of water contamination from untreated sewage via CSO's. The comparison was conducted for exponential DR model only, because it is more grounded in mechanistic considerations of the host-pathogen interaction than the empirical log linear model. Figure 4.4 summarizes the wet weather risk estimates per 1000 users for all the sites using both the indicator organisms (enetrococci risk estimates are obtained from chapter 3, Table-3.8).

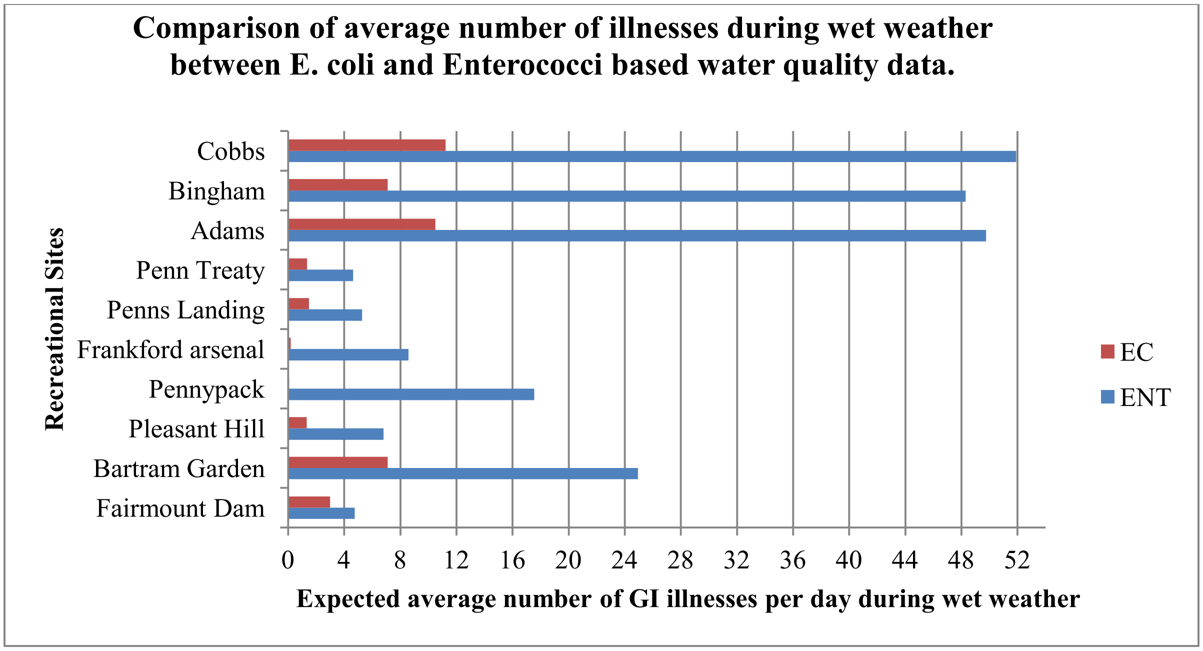

Figure 4-4 Comparison of wet weather mean risk estimates based on E. coli and enteroccoci concentrations.

Comparing the risk of GI illnesses predicted by both the indicators, it appears that there is a significant difference between the number of illnesses predicted by each FIB model. This disparity in numbers was found to be the greatest for sites along the creeks as compared to the river sites. It can be attributed to characteristic difference between the study waters and the water analyzed for the criteria development in 1986. The creeks are shallower with lower flow characteristics than the 1986-epidemiological fresh water bodies, which might have resulted in lower *E. coli* to enterococci ratio causing the model to predict risks differently. This model disparity in risk estimates can influence the watershed management efforts and decision making process significantly as based on *E. coli* model only two sites (COB and AD) presented elevated health hazard however based on enterococci model this number raised to six; including sites from Delaware and Schuylkill river too (PP, FA, BH and BG; Figure 4.4). To be more precise, the differences in risk estimates at the Schuylkill and Creek sites between the two FIB models need more attention. These sites were investigated at the same time (2007), unlike Delaware River sites; which were sampled for *E. coli.* in 2010 and for enterococci in 2011. There is a significant drop in risk estimated from EC model for BG and BH site as compared to the results obtained from ENT model, resulting in compliance of these sites with EPA safe-recreational water criteria. Based on EC data at both the sites the expected number of GI illness is 7 per 1000 users per day, however this number is increased by a factor of 2.5 and 6 at BG and BH respectively when ENT data is used (BG-25 illnesses/1000 users/day and BH-48 illnesses/1000 users/day; Figure-4.4). It should be noted that the recreational health risk criteria of less than 8 illnesses/1000 users/day is based on geometric mean values of the water quality data which were averaged over time rather than the daily values as used in our analysis.

At the Delaware River sites, due to analytical difficulties we could not obtain the ENT data for the same year when EC data were analyzed, which is year 2010. Thus the wet-weather risk estimates from ENT model are based on 2011 sampling results. Furthermore, the total precipitation during the recreational season of 2011 was more than double the total precipitation during 2010; 2011 total precipitation - 27.92" and 2010 total precipitation -13.26" (PennState Climatologist). These conditions indicates that greater number of CSO discharges might have occurred in 2011, causing increase in the indicator organism's concentration and thus leading to increased level of GI illnesses at these sites.

4.4 Conclusion

The risk assessment carried out in this chapter targeted to address the health risk concern of recreational users of Philadelphia and attempted to present a detail perspective about the usage profile and the potential of different non-swimming activities in causing GI illnesses to the exposed population. Fewer sites were found above the criteria limit using EC data than ENT data, this finding is significant since E. *coli* is the preferred indicator of fecal contamination for freshwater sites (U.S. EPA 2011). Adams (AD) and Cobbs (COB), presented elevated risks using both the indicator models and thus require targeted action to control any future health hazard from recreation at these locations. In particular, playing with water and wading were found most prominent and potential activities leading to higher risk of GI illnesses at these sites. These activities are categorized as non-swimming activities and are not usually considered critical in evaluating health risk potential of a water body. But our investigation has shown that under episodic events of poor water quality, such exposure scenarios can also lead to excessive risk of GI illness.

It should be taken into account that the probabilistic risk estimates generated in this study have multiple sources of uncertainties in its projections and thus the results should be interpreted with some caution. The epidemiological studies used to derive the dose-response model referred to the study locations that may differ significantly from those encountered in the present study. Significant differences include much lower bather density at the creek sites than at the sites observed during epidemiological studies, different proximity to wastewater treatment plant outfalls, and likely other features. But the model presented in this study outlines the characteristic difference in the usage potential and associated health risk levels at small creeks and large rivers. Collectively, sites close to an existing CSO’s and located on small creeks need to be examined for wading and playing exposure scenarios, while large river bodies should address on-shore fishing opportunities more cautiously when subjected to a nearby sewage outfall.

5 Chapter 5: Recreational use projection for the Schuylkill and the Delaware River segments, within the City of Philadelphia.

5.1 INTRODUCTION

For sustainable management of the waterway system in Philadelphia, it is important for the public utilities to be able to accurately measure and describe the recreational uses within the City limits. However, the large size of the water bodies and the dispersed nature of recreational resources make it logistically challenging and expensive to collect these much-needed data. Data surveys and sampling procedures are often practiced to quantify recreational uses (Smallwood and Beckley 2008; Fay, Colt et al. 2010). Many researchers have implemented different survey techniques; such as traffic counters, on-site observational surveys or aerial surveys for measuring usage frequency on sampled days and then extrapolated it to the entire study period (Kay, Fleisher et al. 1994; Nelson and Valentine 2002; Soller, Olivieri et al. 2003; Smallwood, Beckley et al. 2011). On the contrary, in the current study continuous dataset at selective sites were gathered throughout the study period; using time-lapse construction cameras (results shown in chapter 2). However, an estimate for usage pattern in the remaining segments of the water-bodies was still missing; which would be a useful tool for the City officials in making informed management decisions.

Furthermore as the overall goal of this dissertation was to evaluate the total health risk associated with water-use in Philadelphia, it was necessary to develop a technique to project the usage across the entire river length within the City. Water recreational use prediction is a process achieved through several techniques and is used to estimate different measurements of the recreation units such as participation level and participation frequency (specific to site) or number of trips and count of days spent in recreation (specific to users). Three basic types of approaches used for predicting recreational participation are- 1) site-specific user approach, 2) site-specific aggregate approach, and 3) population specific approach (Bowker 2001). For first two approaches recreational usage information at the study sites are required, while population-level scenarios are usually household based. For this study, we used a quasi site-specific aggregate approach wherein expert's information was combined with individual site level data to project participation and use by activity. In this analysis our objective was to provide reliable estimates of projected recreational use on both the rivers, via a consistent and statistically valid approach. Hence we developed a method to integrate the camera collected information with the expert opinion of water-resource managers to give baseline estimate of recreational activities along the two main waterway systems in Philadelphia; the Schuylkill River and the Delaware River. Online- survey and interviews of park officials were conducted to collect usage information about the river miles falling under their jurisdiction and mathematical models were developed to relate the survey information to the camera-collected data in order to predict user's frequency in non-monitored segments of the river.

Development of baseline recreation data will not only help the natural resource managers identify changes in use patterns, but will also facilitate them with a tool to forecast future recreational health risk potential from water sport advancements and infrastructure developments on these water bodies.

5.2 DATA COLLECTION

As mentioned earlier in chapter 2, using time-lapse construction cameras comprehensive information was collected for water-based recreational exposure pattern at different locations in

Philadelphia. Out of 10 study sites, 7 were located along two main rivers; the Schuylkill and the Delaware River. These river sites (Schuylkill River – FD & BG; Delaware River – PH, PP, FA, PT &PL) were surveyed during the recreational season of 2010 and a detail summary statistics for users count per wet/dry day by activity was prepared to characterize the usage pattern at each site (results discussed in Chapter 2). In order to expand our estimate from the local monitored segments to the larger stretch of the river, we required inventory of usable river miles within the City. Hence we conducted an online survey to collect the usage information across the entire river-stretch, which allowed us to incorporate riverfront characteristics optimal for recreational use into the results of our study.

For the survey, we divided both rivers into 3 consecutive sections. Each section was further sub-divided in 13 to 15 successive segments of 0.2 to 1.5 miles length. The boundaries for the segments were decided after thorough discussion with the park officials. The criteria followed were-

- For sections which were covered by camera surveillance the boundary coincided with the camera field of view.
- For the remaining stretch of the river, the water reach expected with uniform usage potential was considered as one segment. The usage potential was mainly governed by the land use type of the water-front property.

During the early spring of 2012, we contacted the concerned authorities who knew the river usage and had been involved in the river-front development activities to give us the best estimate of recreational-water activities in each segment. Specifically, the authorities were asked to rate each segment on a scale from 1 to 5 (least likely to most likely) for the possibility of observing

Boating, Boat fishing, Fishing, Jet skiing Kayaking, Swimming, Wading, and Playing with water in that segment. Approximate time required to fill- in each survey ranged from 15 min to 25 min. Each survey had to be completed in one sitting. The officials approached for filling the online survey are listed in Appendix-A, with links to each of the survey forms.

Three surveys for the Schuylkill River were as following–

1) Upper Schuylkill section "Montgomery City line to Falls Bridge" ~4.88 miles– 7segments
2) Middle Schuylkill section "Falls Bridge to Fairmount Dam" ~3.65 miles– 13 segments
3) Lower Schuylkill section "Fairmount Dam to Fort Mifflin" ~9.6 miles – 13 segments

Three surveys for the Delaware River were as following–

1) Upper Delaware section "Poquessing Creek to Allegheny Ave" ~ 8.6 miles- 13 segments
2) Middle Delaware section "Allegheny Ave to Pattison Ave" ~ 6.5 miles – 14 segments
3) Lower Delaware section " Pattison Ave to Fort Mifflin" ~ 6 miles- 8 segments

The organizations which played an instrumental role in collecting the information were – Schuylkill Banks, Schuylkill River Development Corporation, Philadelphia Parks and Recreation, Manayunk Development Corporation, Delaware River City Corporation, Delaware River Waterfront Corporation and Pennsylvania Environmental Council. At first, multiple meetings and seminars were conducted with these city development organizations to educate the target audience about the concept of the study and to develop common interest grounds to get a reasonable output. After the initial meetings, appropriate changes in the segment boundaries were done based on the recommendations made by the authorities. In order to maintain consistency in the way each respondent would analyze the river segments, we outlined certain key features to be considered for answering survey questions. Respondents were requested to evaluate all the river segments for suitability of each activity based on the following criteria -

- Geographic features-amount of beachfront, attractiveness
- Direct access to water for swimming
- Recreational infrastructure development
- Zoning adjacent to water sites
- Amount of continuous waterway for boat travel
- Ease of access (easy to find/signage, adequate parking facilities, accessible by car or restricted; close to major roadway)

These criteria were discussed in every meeting and were also mentioned on the survey template. A sample of the survey template used to collect information is given in Appendix-B and the river segments with boundary details are discussed in Appendix- C. In brief, the survey format included Google Earth images of each segment followed by two sets of questions-

a. Rate the segment from ‘less than 1 in 10 probability’ to ‘greater than 9 in 10 probability’ for the observed activities.
    i. The available options were- less than 1/10, 2/10, 5/10, 8/ 10, greater than 9/10.

b. Provide probability of having new access point on that segment in future – such as fishing pier, jet ski dock, boat dock, kayak dock.
    i. The options available were – less likely, moderately likely, strongly likely.

As stated by other researchers (Porter 2004; Russell, Verhoef et al. 2004; Shooter 2009), our study also faced the similar issue of low response rate from the respondents. Out of three survey-sections per river, we were able to get response for only two sections; with zero response rates for Middle Schuylkill River and Lower Delaware River sections. Due to lack of data, these sections were dropped from our analysis. For other sections, on an average we got three responses for each survey. The second set of the question regarding probability of a future access point per segment was intended to collect information for predicting future usage profile. However due to time constrain and lack of sufficient data we addressed only the spatial projection and documented information that could be useful for making projections in time.

5.3 DATA ANALYSIS AND RESULTS

The survey results were analyzed with an aim to 1) Establish a relationship for each activity between the data observed by cameras and the survey response, and 2) For activities where strong association between two methods is observed, extrapolate that relationship to the rest of the river miles and estimate the respective number of the users for that activity. The data analysis was carried out in two stages. Stage 1- the segments which were covered by the camera-surveillance; from here onwards named as "*monitored segments*" were analyzed for association between survey response and camera data. Stage 2- the summary statistics developed for *monitored segments* was then applied to the *un-monitored segments* (segments not surveyed by camera) to get a camera-equivalent count of users distribution.

5.3.1 Stage 1: Calculations and Results

In order to estimate users count in the *monitored-segments* of the river, following 3 steps were conducted-

1) Calculate the probability of observing an activity in the given segment based on the online survey response, termed as "*survey ranking*". Example calculation-

   a. $$(SurveyRanking)_{per\ activity} = \left[0.5 \times (\%\ Respondent\ agreed\ for\ Prob < \tfrac{1}{10})\right] + \left[2 \times (\%\ Respondent\ agreed\ for\ Prob = \tfrac{2}{10})\right] + \left[5 \times (\%\ Respondent\ agreed\ for\ Prob = \tfrac{5}{10})\right] + \left[8 \times (\%\ Respondent\ agreed\ for\ Prob = \tfrac{8}{10})\right] + \left[9.5 \times (\%\ Respondent\ agreed\ for\ Prob > \tfrac{9}{10})\right]$$

2) Develop probability distribution function (PDF) for the survey response.
3) Normalize the survey PDF with respect to the total number of users observed by camera for that segment to get survey-predicted users count for each activity.
4) Perform linear regression analysis for each activity to model relationship between the camera-observed user count (dependent variable) and survey-predicted user count (explanatory variable).

The sample calculation for the first segment of Lower Schuylkill River section is shown in Appendix-D. For both the rivers, the maximum data points available to develop linear regression models were 4. It was hard to detect nonnormality with small number of data points and so the population distribution was assumed normal in order to provide a baseline estimate on usage frequency.

5.3.1.1 Schuylkill River Survey summary Fig 5.1 and Fig. 5.2, represents the comparison between the survey-predicted users count and the camera-observed no. of users for all the activities at the monitored-segments of the Schuylkill River. Only two sites (Fairmount Dam (FD) and Bartram Garden (BG)) were monitored with camera on the Schuylkill River. Since it is always required to have more observations than parameters; for linear regression we needed one more data point. Hence, telephonic interviews were conducted with Schuylkill Navy volunteer group and boat house organizers to gather estimates on users count in one of the segment of the upper Schuylkill River; (Montgomery County line to Flat Rock Dam). This segment was named as "city" in the plots (Fig. 5.1 and Fig. 5.2).

For major uses on the Schuylkill River i.e. kayaking and fishing, we observed very good estimate on users count using the camera surveillance (R^2=0.99, p-value=0.04; R^2=0.94, p-value= 0.1 respectively), while other uses were slightly under-estimated with not a statistically significant correlation. This is understandable as due to fixed field of view by camera, these transient activities (such as- boating, jet skiing) could not be accounted with accuracy in camera surveillance. However the relationships developed for boating, boat-fishing and jet skiing (Fig. 5.1) were not significant, they were strong enough to be considered for predicting usage in un-monitored segments.

5.3.1.2 Delaware River Survey summary The survey results for Delaware River sites were examined with a slight modification. Due to large numbers of row clubs and boat houses located on the Delaware River, there was huge variation in boating and kayaking usage potential between river segments. Hence, the predictions obtained based on the linear models developed for monitored-segments could not be applied to the un-monitored segments. Therefore, we dropped boating and kayaking from the survey analysis and normalized all the camera observations without these activities to examine the relation with the survey estimates. Fig 5.3 and fig. 5.4 – represents the comparison between the survey-predicted users count and the camera-observed no. of users for each activity at all the monitored sites along the Delaware River.

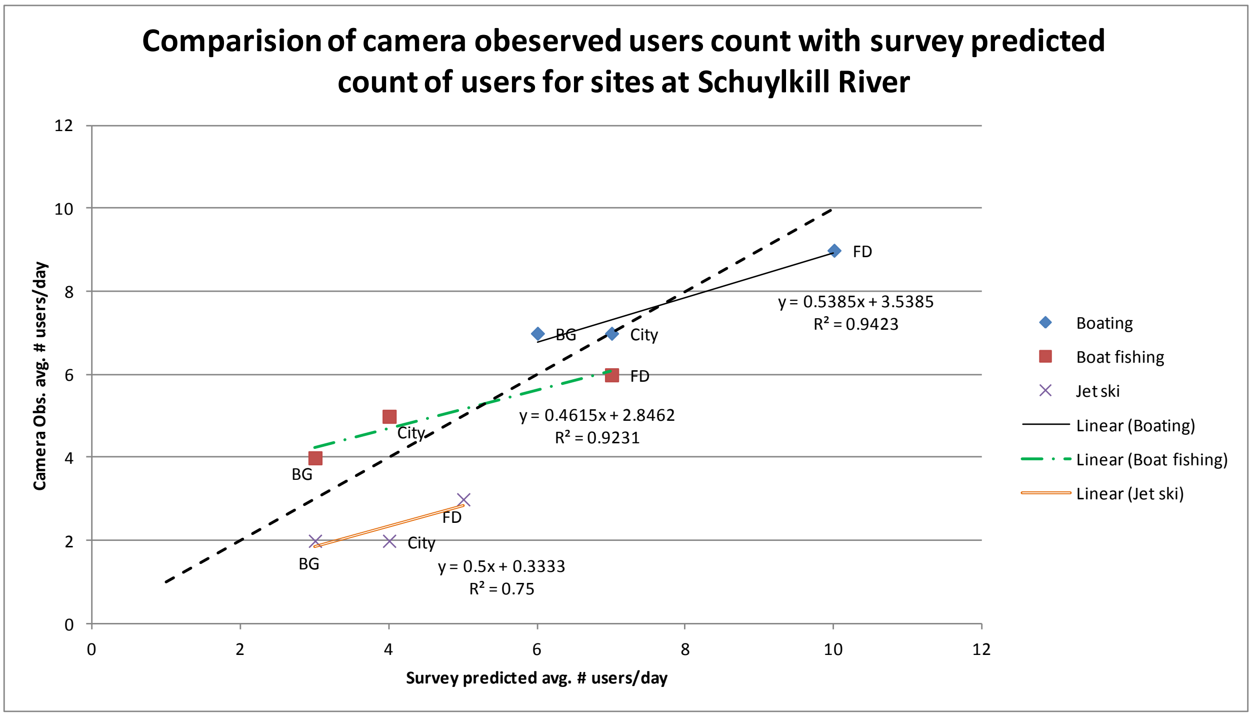

Figure 5-1 Comparison of camera observed users count and survey predicted count of users for common activities on the Schuylkill River.

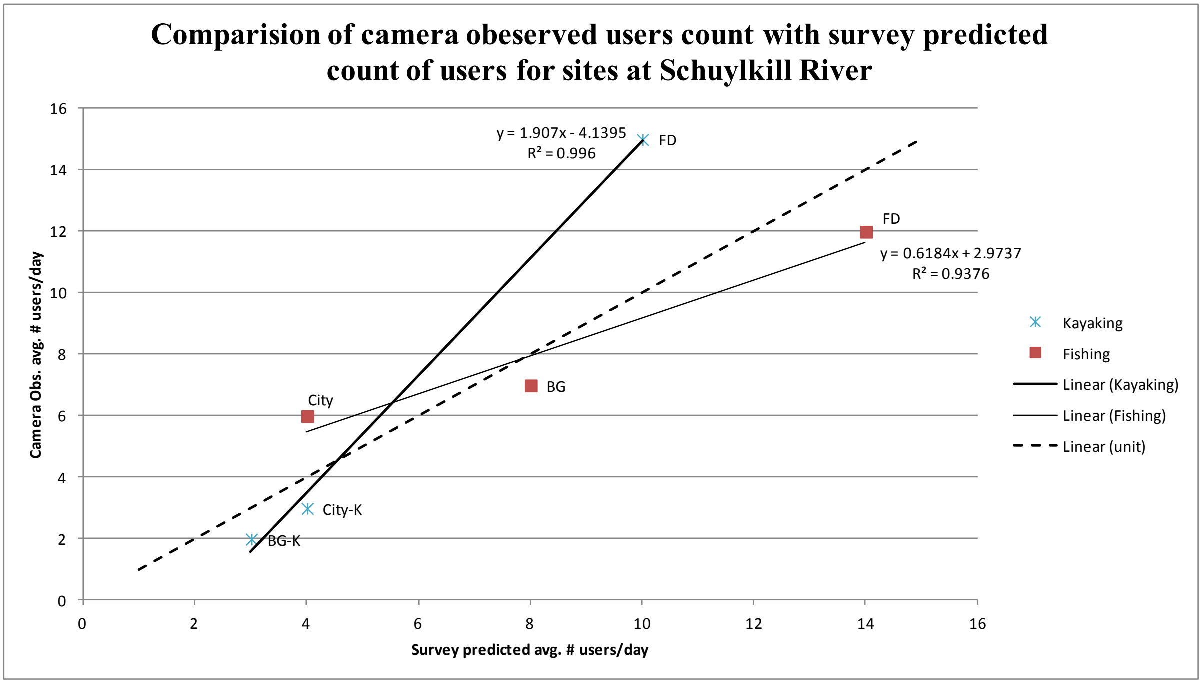

Figure 5-2 Comparison of camera observed users count and survey predicted count of users for major water-use activities at Schuylkill River.

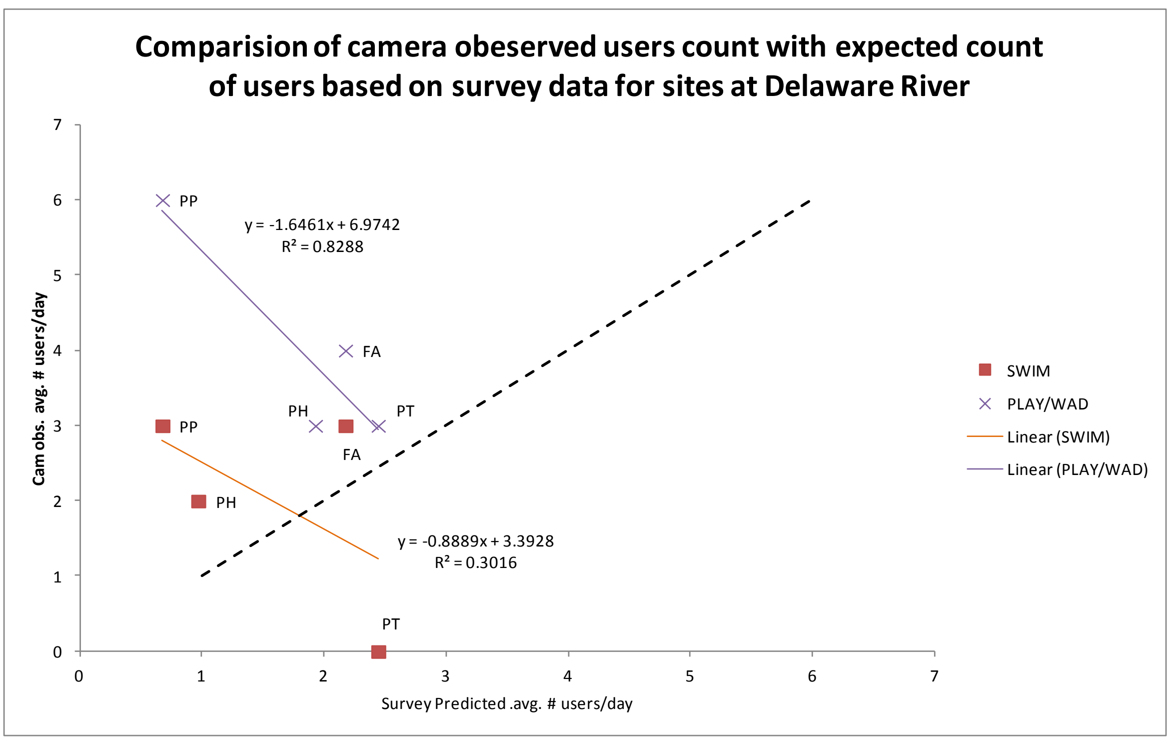

Figure 5-3 Comparison of camera observed users count and survey predicted count of users for potential non-transient activities on Delaware River.

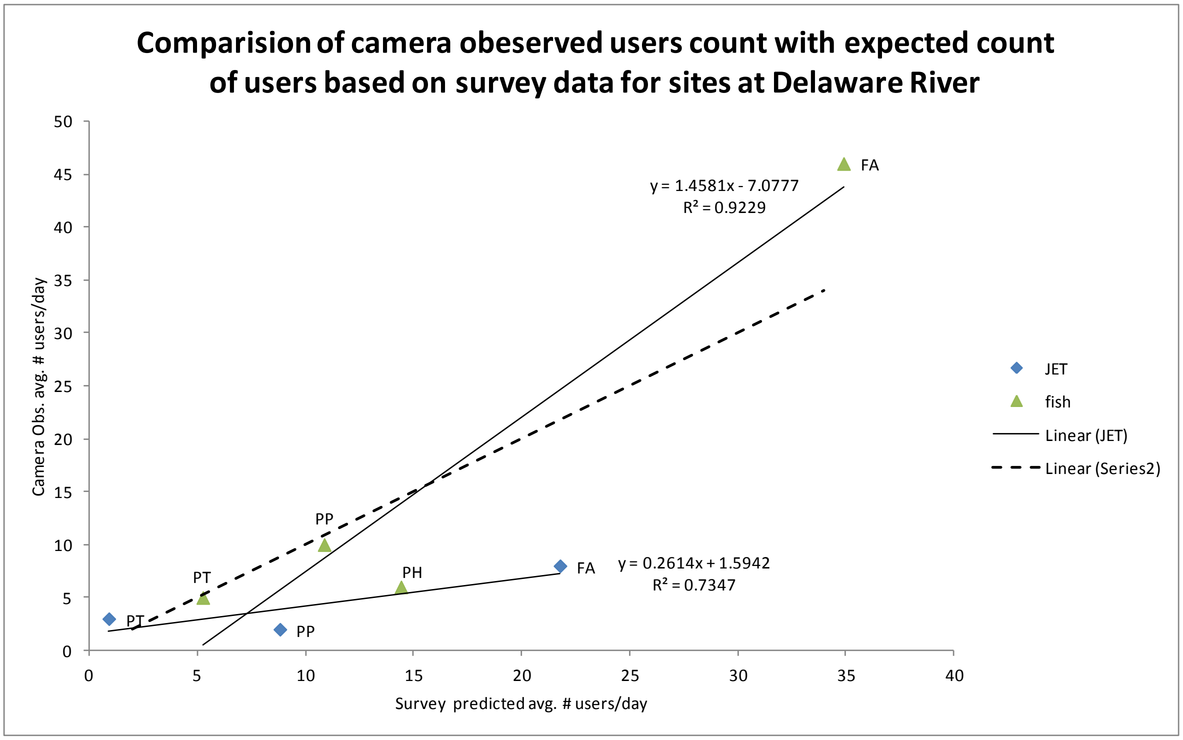

Figure 5-4 Comparison of camera observed users count and survey predicted count of users for fishing and jet skiing on Delaware River.

It was interesting to observe that non-transient activities especially those involving greater water-ingestion potential such as- swimming and playing/wading were largely under estimated by the survey respondents (Fig-5.3). However, there was a good statistically significant correlation observed between the camera data and survey predicted data for on shore-fishing, which is the second major activity on the Delaware River after boating (R^2= 0.92, p-value=0.04; Fig-5.4). The model for predicting playing frequency was also found statistically significant with p-value as 0.05 (R^2=0.83, Fig. 5.3). However the relationships developed for swimming and jet skiing were not significant, in absence of additional data they were considered appropriate to be used for further analysis.

5.3.2 Stage 2: Calculations and Results

In order to estimate users count in *un-monitored segments* of the rivers, following steps were conducted:-

1) Analysis for *monitored-segments*:- Both the survey-ranking per activity and the camera-observed user count per activity were added across all uses for each segment (Table 5.1 and Table 5.2).
2) Analysis for *un-monitored segments*:- The *camera-equivalent total count of users* corresponding to *the survey ranking total* was linearly extrapolated assuming that the ratio of survey- ranking total to camera-observed total counts remains same across all the segments (monitored and un-monitored segments). The obtained results are summarized in Table 5.3 and Table 5.4 for the Schuylkill and the Delaware River, respectively. The formula used for this calculation is represented below -
    a. $\left(\text{Camera}_{\text{equivalent}} Total\ Count\right)_{\text{Un}-\text{monitored seg.}} = \dfrac{\sum_{i=1}^{n}\left\{\dfrac{\left(\text{Camera}_{\text{observed}} Total\ Count\right)_{\text{monitored seg.}}}{(Survey\ Ranking\ Total)_{monitored\ seg.}}\right\}}{n} \times (Survey\ Ranking\ Total)_{un-monitored\ seg.}$

    Where n is no. of monitored segments per river; for Schuylkill River, n=3 and for Delaware River, n=4.
3) Prediction of users per activity in each *un-monitored segment*:- Normalized the survey PDF with respect to the Camera-equivalent total count for that segment to get survey-equivalent users count for each activity.
    a. The survey-equivalent user's count for each activity was then applied to the developed linear models to predict the camera-equivalent user's count per activity in each *un-monitored segment*. The results are summarized in Table 5.5 and Table 5.6 for the Schuylkill and the Delaware River, respectively.

Table 5-1 Schuylkill River survey summary for *monitored-segments*

Survey section of Schuylkill River	Camera monitored sites	Survey ranking (responses added across all uses in each segment)	Camera observed total users
Lower SRT	seg 1 (FD)	13.7	45
Lower SRT	seg 8 (BG)	11.5	22
Upper SRT	seg 1 (City)	20.2	25

Table 5-2 Delaware River survey summary for *monitored-segments*

Survey section of Delaware River	Camera monitored sites	Survey ranking (responses added across all uses in each segment)	Camera observed total users
Upper Del	seg 2 (PH)	13.9	27
Upper Del	seg 5(PP)	15.5	21
Upper Del	seg 10 (FA)	14	61
Middle Del	seg 5 (PT)	15.6	11

Table 5-3 Schuylkill River camera-equivalent total counts of users for *un-monitored-segments*

Survey section of Schuylkill River	Un-monitored segments	Survey ranking (responses added across all uses in each segment)	Camera-equivalent total count
Upper SRT	Seg 2	8.1	17
	Seg 3	12.2	26
	Seg 4	13.7	29
	Seg 5	**27**	58
	Seg 6	**21.6**	46
	Seg 7	**24**	51
Lower SRT	Seg 2	11.4	24
	Seg 3	12.4	27
	Seg 4	10.7	23
	Seg 5	7.2	15
	Seg 6	10.4	22
	Seg 7	10.2	22
	Seg 9	6	13
	Seg 10	7.5	16
	Seg 11	6.5	14
	Seg 12	5.5	12
	Seg 13	10.4	22

Table 5-4 Delaware River camera-equivalent total counts of users for *un-monitored-segments*

Survey section of Delaware River	Un-monitored segments	Survey ranking (responses added across all uses in each segment)	Camera-equivalent total count
Upper Del	Seg 1	12	25
	Seg 3	4	9
	Seg 4	12	26
	Seg 6	8	16
	Seg 7	9	18
	Seg 8	13	27
	Seg 9	13	28
	Seg 11	9	20
	Seg 12	5	10
	Seg 13	12	25
Middle Del	Seg 1	7	15
	Seg 2	5	10
	Seg 3	3	6
	Seg 4	3	6
	Seg 6	4	9
	Seg 7	11	22
	Seg 8	11	23
	Seg 9	14	29
	Seg 10	10	22
	Seg 11	4	9
	Seg 12	10	21
	Seg 13	8	16
	Seg 14	2	4

It was found that on Delaware River with respect to non-transient activities ; we covered all the potential sites using camera surveillance because all the survey rankings in un-monitored segments (Table 5.4) were less than the average ranking of the monitored segments (Table 5.2; Average ranking=14.8). However, for the Schuylkill River we observed greater usage potential at segments- 5, 6 and 7 (highlighted with bold; Table 5.3) than the segments monitored by the camera (Table 5.1; Average ranking=15.1). This finding enhances our confidence in the estimated results for other un-monitored segments as it reflects the real scenario. There is a Philadelphia Canoe Club on this section of the river which participates in seasonal regatta races; resulting in greater usage ranking for these segments.

The predicted camera-equivalent user's count per activity per segment as derived from the linear models represents the relative usage potential of a given river-segment within the City (Table 5.4 & Table 5.6). Based on the Schuylkill River results it can be estimated that boating is approximately equally preferred all across the river; while kayakers are less likely to navigate south of Bartram Garden. Similarly, fishing was found as a common activity with pockets of strong usage demand around Fairmount Dam. Jet skiing was not estimated as an intensive activity but it is expected to occur all across the river (Table 5.5).

Table 5-5 Distributed users count for a given day during the recreational season with respect to each activity on the Schuylkill River (highlighted columns represent the segments monitored by camera).

Rec. Uses on Schuylkill River	Upper Schuylkill							Lower Schuylkill												
	City							FD							BG					
	Seg 1	Seg 2	seg 3	Seg 4	Seg 5	Seg 6	Seg 7	Seg 1	Seg 2	Seg 3	Seg 4	Seg 5	Seg 6	Seg 7	Seg 8	Seg 9	Seg 10	Seg 11	Seg 12	Seg 13
Boating	7	5	6	6	11	11	11	9	7	7	7	7	7	7	7	6	8	8	6	9
Boat fishnig	5	4	5	4	9	7	7	6	4	4	4	4	4	4	4	3	3	3	3	5
Fishing	6	7	7	7	11	8	11	12	7	8	7	4	8	8	7	5	3	4	4	4
Jet ski	4	2	1	1	1	0	0	3	2	2	2	2	2	1	2	1	3	1	1	2
Kayak	3	4	10	18	28	24	24	15	8	8	4	0	0	0	2	0	0	0	0	0

Table 5-6 Distributed users count for a given day during the recreational season with respect to each activity on the Delaware River (highlighted columns represent the segments monitored by camera).

Rec. Uses On Delaware River	Upper Del												
		PH			PP					FA			
	seg 1	seg 2	seg 3	seg 4	seg 5	seg 6	seg 7	seg 8	seg 9	seg 10	seg 11	seg 12	seg 13
Fishing	7	4	0	13	10	0	5	14	15	46	3	0	20
Jet Ski	5	18	3	4	2	4	3	4	4	8	4	3	3
Swimming	2	2	2	2	3	2	2	2	2	3	2	2	2
Wad/Play	5	3	5	5	6	5	5	5	5	4	5	5	5

Rec. Uses On Delaware River	Middle Del													
					PT				PL					
	seg 1	seg 2	seg 3	seg 4	seg 5	seg 6	seg 7	seg 8	seg 9	seg 10	seg 11	seg 12	seg 13	seg 14
Fishing	8	1	0	0	5	0	5	8	13	8	0	10	10	0
Jet Ski	2	2	2	2	3	3	3	4	5	4	2	2	2	2
Swimming	2	2	2	2	0	2	2	2	2	2	2	2	2	2
Wad/Play	5	5	5	5	3	5	0	5	5	5	5	0	5	5

The Delaware River usage distribution, as shown in Table 5.6, represents that as compared to the middle Delaware section; fishing is more prominent in upper Delaware River sections. As discussed earlier, survey respondents associated very low probability with swimming and wading/playing along the river which resulted in survey –equivalent count of 1 user at max. When the survey-equivalent estimate of 1 user was applied to the linear model it gave the constant camera equivalent estimate for each activity (2 users for swimming and 5 users for wading/playing). This is essentially a model prediction error at low value ranges, but results in conservative estimates which can be corrected later by adding more monitoring sites along the Delaware River.

5.4 CONCLUSION

A baseline characterization of current recreational use pattern on the Schuylkill and the Delaware River within the City limit was presented in this chapter. Due to developing opportunities for water-exposure along these rivers, it was essential to have an estimate about recreational preferences of users at a much broader scale. As a result, it was found that fishing which was amongst the most frequent activities at the monitored locations; was consistently preferred over other activities across a large percentage of the river- mile. For Delaware River, the estimates are not that conclusive because we were not able to include boating and kayaking in our analysis. It should be noted that due to canoe clubs and boat houses on the Schuylkill River and Delaware River respectively, our estimates for boating/kayaking may not be accurate enough to predict actual count of users on daily basis on these water bodies. Additional data from individual rowing/boating associations about their member counts and seasonal events may help us in giving better estimates of these activities.

This information is intended to assist PWD by providing them a framework for development of additional data evaluation procedures. These estimates may also improve the ability of water resource managers to measure recreational use on river systems by providing them with a benchmark for long-term monitoring.

6 CHAPTER 6: PROJECTED RISK OF GASTROINTESTINAL ILLNESSES DUE TO RECREATIONAL EXPOSURES ACROSS THE SCHUYLKILL AND THE DELAWARE RIVER

6.1 Introduction

The recreational use pattern along the two rivers in Philadelphia was analyzed by conducting online surveys and camera surveillances. In this chapter we integrated that information with other exposure factors to expand the risk estimates to un-monitored river segments. The QMRA approach as we discussed in previous chapters was applied to the recreational use data presented in chapter 5 and the risk estimates were analyzed to identify the relative risk potential of different activities along the studied river sections. In order to conduct the analysis several assumptions were made to extrapolate the data from dispersed sampling locations to the entire river stretch. The aim of this exercise was to present the first cut risk estimates for the river miles within the city based on the current usage pattern. We have identified earlier that both the major water bodies of Philadelphia are heavily used for recreational purposes and are also subjected to wet weather discharges. Hence, the overall risk estimate will provide a basic framework for assessing the health impact of different recreational activities throughout the river and the developed tool can be further expanded to conduct comprehensive analysis at other water bodies in the region.

6.2 Data analysis

6.2.1 Recreational use assessment

The number of users in each segment of the river was estimated by developing a mathematical relationship between the online survey results and the camera observations. The calculation approach and the final results of recreational use distribution by activity at each river segment was presented in the previous chapter (Chapter 5, Table 5.2 and 5.3) and also summarized in Table 6.1 and 6.2 for the Schuylkill and the Delaware River respectively.

Table 6-1 Distributed users count for a given day during the recreational season with respect to each activity on the Schuylkill River (highlighted columns represent the segments monitored by camera).

Rec. Uses on Schuylkill River	Upper Schuylkill							Lower Schuylkill												
	City							FD							BG					
	Seg 1	Seg 2	seg 3	Seg 4	Seg 5	Seg 6	Seg 7	Seg 8	Seg 9	Seg 10	Seg 11	Seg 12	Seg 13	Seg 14	Seg 15	Seg 16	Seg 17	Seg 18	Seg 19	Seg 20
Boating	7	5	6	6	11	11	11	9	7	7	7	7	7	7	7	6	8	8	6	9
Boat fishnig	5	4	5	4	9	7	7	6	4	4	4	4	4	4	4	3	3	3	3	5
Fishing	6	7	7	7	11	8	11	12	7	8	7	4	8	8	7	5	3	4	4	4
Jet ski	4	2	1	1	1	0	0	3	2	2	2	2	2	1	2	1	3	1	1	2
Kayak	3	4	10	18	28	24	24	15	8	8	4	0	0	0	2	0	0	0	0	0

Table 6-2 Distributed users count for a given day during the recreational season with respect to each activity on the Delaware River.

Rec. Uses On Delaware River	Upper Del												
		PH			PP					FA			
	seg 1	seg 2	seg 3	seg 4	seg 5	seg 6	seg 7	seg 8	seg 9	seg 10	seg 11	seg 12	seg 13
Fishing	7	4	0	13	10	0	5	14	15	46	3	0	20
Jet Ski	5	18	3	4	2	4	3	4	4	8	4	3	3
Swimming	2	2	2	2	3	2	2	2	2	3	2	2	2
Wad/Play	5	3	5	5	6	5	5	5	5	4	5	5	5

Rec. Uses On Delaware River	Middle Del													
					PT				PL					
	seg 14	seg 15	seg 16	seg 17	seg 18	seg 19	seg 20	seg 21	seg 22	seg 23	seg 24	seg 25	seg 26	seg 27
Fishing	8	1	0	0	5	0	5	8	13	8	0	10	10	0
Jet Ski	2	2	2	2	3	3	3	4	5	4	2	2	2	2
Swimming	2	2	2	2	0	2	2	2	2	2	2	2	2	2
Wad/Play	5	5	5	5	3	5	0	5	5	5	5	0	5	5

6.2.2 Exposure assessment

To conduct exposure assessment, information about the ingested dose and total count of users engaged in each activity was required at both the rivers. Only two segments at Schuylkill River and 5 segments at Delaware River were monitored for bacterial information. Thus pooling test on wet weather enterococci data was done to verify if data from any two segments on the same river can be combined to represent the water quality between those segments. None of the segment combination passed the pooling test, thus the river stretch was divided in consecutive zones where each zone had at least one water sampling site. As such, the Schuylkill River was divided in 3 zones and the Delaware River was divided in 5 zones (Table 6.3). It was assumed that the sampled site represented the water quality of all the segments falling in that particular zone. For zone 1 on the Schuylkill River since there was no water sample data, the Fairmount Dam (FD) and Bartram's Garden (BG) wet weather enterococci data were combined to give best estimate of water quality in this region. The zone boundaries were defined based on the stretch of the river that is commonly used for continuous boating and these boundaries were also discussed with the park officials for confirmation.

Table 6-3 Zone characterization for exposure assessment

River Segments	Schuylkill River Zone	Schuylkill River Water Quality	Delaware River Zone	Delaware River Water Quality
seg-1	Zone 1	Pooled FD and BG data	Zone 1	PH
seg-2				
seg-3				
seg-4			Zone 2	PP
seg-5				
seg-6				
seg-7				
seg-8	Zone 2	FD data	Zone 3	FA
seg-9				
seg-10				
seg-11				
seg-12				
seg-13				
seg-14			Zone 4	PT
seg-15	Zone 3	BG data		
seg-16				
seg-17				
seg-18				
seg-19				
seg-20			Zone 5	PL
seg-21	-	-		
seg-22	-	-		
seg-23	-	-		
seg-24	-	-		
seg-25	-	-		
seg-26	-	-		
seg-27	-	-		

6.2.2.1Schuylkill River For calculating the total user count per zone, all the non transient activities were added across the segments while for transient activities maximum users count in a given stretch was considered (Table 6.4). As an exception, zone 1 was further divided into 3 sub-zones due to typical physical characteristics of the river in this zone. The max user count in each sub-zone was added to get the final users count in this zone. Sub-zones for zone 1-

i) Sub-zone –i: included only segment 1(upstream of Flat Rock Dam)

ii) Sub-zone –ii: included segments 2, 3 (parallel to Manayunk canal)

iii) Sub-zone –iii: include segment 4, 5, 6, 7 (regatta corridor)

Based on users count presented in table 6.1, the example calculation for Boaters in zone 1-

$$= \max(\text{sub-zone})_{i} + \max(\text{sub-zone})_{ii} + \max(\text{sub zone})_{iii}$$

$$= 7+6+11 = 24 \text{ boaters in zone 1 per day}$$

6.2.2.2 Delaware River While interpolating users count from survey responses, the large number of row clubs and boat houses on this river made it difficult to estimate boaters' number for un-monitored river segments. Thus for the Delaware River, the user distribution statistics was prepared for non-boating activities as shown in Table 6.2. Similar approach as used for Schuylkill River zones was applied for the Delaware River zones to calculate the total users count per transient and non-transient activity. The final result obtained for the Delaware River zones is shown in Table 6.5.

Table 6-4 Total number of users per zone for Schuylkill River on a given day.

Total Users On SRT-		zone 1		zone 2		zone 3	
		SUM	**Max**	**SUM**	**Max**	**SUM**	**Max**
Transient	Boating	-	24	-	7	-	9
	Boat fishnig	-	19	-	6	-	5
	Jet ski/Kayak	-	48	-	20	-	4
Non-transient	Fishing	57	-	54	-	27	-

Table 6-5 Users profile per zone for Delaware River on a given day.

Total Uses On Delaware		zone 1		zone 2		zone 3		zone4		zone5	
		SUM	Max	SUM	Max	SUM	Max	SUM	Max	SUM	Max
Non-transient	Fishing	11	-	28	-	99	-	14	-	53	-
	Swimming	7	-	10	-	15	-	12	-	20	-
	Wad/Play	13	-	22	-	30	-	29	-	31	-
Transient	Jet Ski	-	18	-	4	-	8	-	3	-	7

6.2.3 Risk Assessment

The risk of GI illness from recreation in these water bodies was calculated by estimating the ingested dose for each activity per zone and then applying it to the exponential dose-response model (Equation 2, Chapter 3).

6.2.3.1Schuylkill River Ingestion rate and exposure duration statistics used for estimating the exposure dose for each activity are presented in Table 6.6. The exposure durations for transient activities were based on the professional judgments provided by the river development council personnel and represented the total amount of time spent by people boating and jet skiing in the river. Recreators travelling in one particular zone were also assumed to spent same amount of time in water as they would spend when travelling all across the river.

Table 6-6 Summary of ingestion volumes and duration estimates used in the Schuylkill River risk assessment.

Observed Activity	Fitted Distribution (Mean, SD)	
	Ingestion Rate (ml/hr)	Exposure Duration (hrs/da)
Jet skiing , Canoeing, Kayaking	LN(5,5) + fixed intake (4 ml/hr)[a]	Tri(1, 2,6)
Boating: power boat, water taxi, tour boat, Ferry	LN(1,0.5)+ fixed intake (1ml/hr)	Tri(2,3,6)
Fishing (on and off shore)	LN(3,2) + fixed intake (1ml/hr)	G(1.3,2.0)
Wading	LN(10,5)[b]	N.O.[d]
Playing with Water	LN(12,6)[c]	N.O.
Swimming, tubing, water skiing	Adults: N(25,5)	N.O.

[a]The fixed ingestion rate accounts for inhalation via mist and droplets in nose and mouth

[b]EPA (2000) recommends an average IR of 10ml/hr for adults, we assumed an SD of 5 ml/hr to create the distribution.

[c]EPA (2000) recommends an average IR of 10ml/hr for wading. We assumed an additional 2ml/hr for playing in the water and an SD of 6 ml/hr.

[d] N.O. stands for Not Observed

6.2.3.2 Delaware River As mentioned above, boating at Delaware River was excluded while interpolating the survey responses. But since it is the most common activity on this water body; different methods were evaluated to incorporate it in the risk estimates. Out of all observed activities, the extrapolation of transient activities duration from monitored segments to the full span of a zone was of main concern at this water body. This is because the duration estimates for the transient activities (boating and jet skiing) were obtained by adding the number of instances when these activities were seen on a particular site on a given day and multiplying it with the interval between the camera images. For example: suppose at Pleasant Hill site, the total number of instances when boating was seen on a given day is represented by “m” and the time interval between images was 2 minutes, then the exposure duration estimated for this site was “2m”. Thus it represented the person hour of use of the activity at a particular site, which makes it specific to the camera-monitored segment only. However, due to the mobile nature of these activities the users are actually traversing across the segments within a zone. Thus for estimating total exposure time the duration spent in other segments should also be added to the duration observed in camera monitored segment. For non-transient activities the duration estimated in monitored segments was assumed to represent the general duration for the recreators in the corresponding zone. The duration distributions for non-transient activities based on camera observations are summarized in Table 6.7; these were applied directly to the corresponding zones to calculate the exposure dose per activity.

Table 6-7 Person Hour use for non transient activities along the Delaware River based on camera observations, Modeled Distribution (parameters)[c] (hrs/day).

Non transient Activities	Site PH[a]	Site PP[a]	Site FA[a]	Site PL[a]	PT[b] (mean)
Wading	N/A	Wei(0.2,0.73)	N/A	N/A	N/A
Fishing	Wei(2.32,1.61)		G(1.09,1.56)	N/A	1.59
Playing with water	LN(0.14,0.31)	Wei(0.31,0.89)	Wei(0.05, 3.01))	N/A	0.17
Tubing/Swimming	LN(0.07,0.04)	G(0.39,1,12)	Wei(0.05, 1.87))	N/A	N/A

[a] Duration of transient activities represents the person hour of use of a particular activity at a site.

[b]In absence of camera observations at PT, duration statistics represents the average values, based on 15 in- person surveys, each of 3 hour duration.

[c] G-Gamma(scale,shape); LN-Log normal(mean, SD); Wei-Weibull(scale,shape).

In addition to exposure duration, the boaters count reported at camera-monitored sites was of concern too. Due to difficulty in distinguishing between images, there was possibility of repetition in boaters and jet skiers' counts. Thus, number of boaters counted at each monitoring segment could be over representing the exposure. In order to address these two sources of error; following different approaches were used to calculate the final risk potential in each zone.

a) Approach 1:- The count of boaters was available for the camera monitored segments, thus we had 5 locations on Delaware River (PH, PP, FA, PT and PL) with duration and frequency data. Under first approach, it was assumed that same number of boaters as seen in monitored segment would travel across the other segments within the particular zone. It was also assumed that the person hour of use for boating/jet skiing in other segments would be same as that observed in the camera-monitored segment. Therefore, the total duration for exposure per zone would be = n * (person hour of use in camera monitored segment). Where "n" represents the number of segment in a particular zone. For example: Zone 1 on the Delaware River –

- Three segments (Seg 1, Seg 2 and Seg 3) are included in zone 1, thus n=3.
- The average person hour of use for boating based on camera observation at the monitored segment (PH, Seg-2) was 48 minutes/day.
- Thus, average time spent boating in zone 1: 3* 48 = 2.4hours/day.

Applying the above method the total exposure duration for boater and jet skiers in each zone was calculated. The results are shown in Table 6.8. This table also gives information about the average boaters recorded at each monitoring segment using camera surveillance, which was combined with the Table 6.5 data to calculate proportional distribution of users by activities.

Table 6-8 Exposure statistics for zones on the Delaware River based on approach 1

Recreational zones	Average Exposure duration		Boaters count/day
	Boating (hrs)	Jet sking (min)	(Monitored segment)
Zone 1	2.4	56	115 (Seg-2, PH)
Zone 2	1.52	14.4	39 (Seg-5, PP)
Zone 3	6.4	43	110 (Seg-10, FA)
Zone 4	4.4	60.1	36 (Seg-18, PT)
Zone 5[a]	Tri(2,6,8)	Tri(1,3,4)	179 (Seg-22, PL)

[a] Zone 5 included Penns Landing Marina region and the duration estimates were based on the telephonic interview with Marina Director at DRWC.

b) Approach 2:- Under this approach we excluded boating and did the tier 1- risk calculations for all other activities using the observed frequency of users per activity. In order to compensate for boating risk potential, the tier1-risk was expressed per 800 users; assuming that at max. 200 boaters per day would traverse across the entire stretch of river with duration distribution as Tri (4, 8,10) hrs/day. The risk per 200 boaters was calculated by applying the average enterococci densities measured across all the five monitored sites. The two risk estimates were added to get the overall risk per 1000 users. 200 boaters were assumed as that is the maximum count of boaters observed in a given day on Delaware River and the exposure duration statistics were obtained from a telephonic interview with Ms Monica Santora (Marina Director, Delaware River Waterfront Corporation). According to Ms Santora, the boaters that dock from Penns Landing tend to stay in river for as long as 10 hours and usually travel large distances. Thus it is possible for them to cross the zone boundaries assumed in our risk analysis.

c) Approach 3:- This approach is very similar to approach 1 except under this approach we did not do the exposure duration correction. Thus, the person hour of use measured at each monitored segment based on camera observations was assumed to represent the exposure duration for transient activities in the corresponding zone. This approach was tested against approach 1. The drawback of approach 1 is that it linearly relates the exposure duration to the number of segments present in each zone. So continuing with the same example of zone 1, if instead of 3 segments this zone was divided in 4 or 5 segments then the exposure duration for boating would have changed from 2.4 hours to 3.2 and 4 hours respectively. Table 6.9 represent the exposure duration statistics used for calculating risk using approach 3, where the duration represents the person hour of use of a particular activity.

Table 6-9 Summary of person hour of use of both transient activities per zone obtained from camera data

Recreational zones	Exposure duration Distributions[b] (hrs/day)		Segment monitored by camera (Site Name)
	Boating	Jet sking	
Zone 1	LN(0.8,1.45)	LN(0.31,0.52)	Seg-2 (PH)
Zone 2	Exp(2.6)	Wei(0.06,1.59)	Seg-5 (PP)
Zone 3	Wei(1.1,1.09)	Wei(0.13,1.17)	Seg-10 (FA)
Zone 4[a]	0.733	0.167	Seg-18 (PT)
Zone 5	B(0.3,0.7,4.1,8.5)	LN(0.13,0.11)	Seg-22 (PL)

[a] In absence of camera observations at PT, duration statistics represents the average values, based on 15 in- person surveys, each of 3 hour duration.

[b] LN-Log normal(mean, SD); B-Beta(min,alpha,beta,max); Wei-Weibull(scale,shape); Exp-Exponential(rate).

6.3 Results and discussion

6.3.1 Schuylkill River

The wet weather risk estimates based on enterococci counts for recreational exposures on the Schuylkill River are presented in Table 6.10. This analysis could not cover the exposure pattern and the associated risk potential across the middle Schuylkill River section: Falls Bridge to Fairmount Dam; which is almost 3.65 miles long. Due to lack of survey response we were not able to develop the input statistics for this section of the river. However almost 80% of the river miles running across the City were evaluated in this exercise and the results collectively showed that the on-shore fishing was the most potential activity leading to maximum contribution in the total health risk estimates on this river (Table 6.10). In the upper reach of the river, the historic Boathouse Row is known for hosting a large number of regattas and boat races, hence kayaking was found as prominent activity in this region and it contributed significantly in the overall risk estimates.

Table 6-10 Average daily estimated cases of GI illnesses and contribution to total risk by each activity for Schuylkill River

Schuylkill River Zones		Zone 1	Zone 2	Zone 3
Avg. risk/1000 users/day (95% CI)		18.22 (1.7 – 59.9)	4.36 (1.66- 11.0)	25.97 (1.8– 95.2)
Avg. risk/ activity/ day	*Boating*	*1.9*	*0.25*	*3.99*
	Boat fishing	*2.3*	*0.31*	*3.45*
	Jet/kayak	*8.6*	*1.57*	*4.07*
	Fishing	*5.3*	*2.24*	*14.46*

The results presented in Table 6.10 further indicated that downstream of Bartram Garden (Zone 3) the health risk projections are relatively higher with an average expected number of GI cases reaching to 26 illnesses/1000 users/day. This is partially because zone 3 is the longest zone covering almost 6 miles as compared to other zones (zone 1 ~ 4.9 miles and zone 2 ~3.7 miles). Also, survey respondents voted for large water traffic in this zone due to scenic boating route up the Schuylkill River from the Delaware River. Further analysis of water quality and the usage pattern in this zone is required to develop a better understanding of the exposure scenarios.

6.3.2 Delaware River

Similar to the results of Schuylkill River survey, response rate for the lower section of the Delaware River survey was negligible. Hence, the region between Pattison Avenue and Fort Mifflin (almost 6 river miles) was excluded from the risk calculation. Furthermore, three different approaches were used to summarize the risk profile for recreation due to discrepancies in measuring boating exposure on this water body.

The average risk of GI illness with 95% confidence interval range was compared for all the three approaches in Figure 6.1. Furthermore, the relative contribution from each activity to the total rate of illnesses per zone is summarized in Table 6.11. Across all the zones, the risk levels were found significantly elevated for zone 2, which covered the Pennypack park region (Figure 6.1; Table 6.11). This finding is significant as based on the recreational use survey zone 2 was rated highest for direct contact activities by the respondents and our analysis also reflected significantly high risk estimates for this zone. Presence of particular geographical features such as direct access to water, close parking facility, multiple fishing spots and large beach front; facilitated increased participation of users in fishing, playing and swimming as compared to other zones.

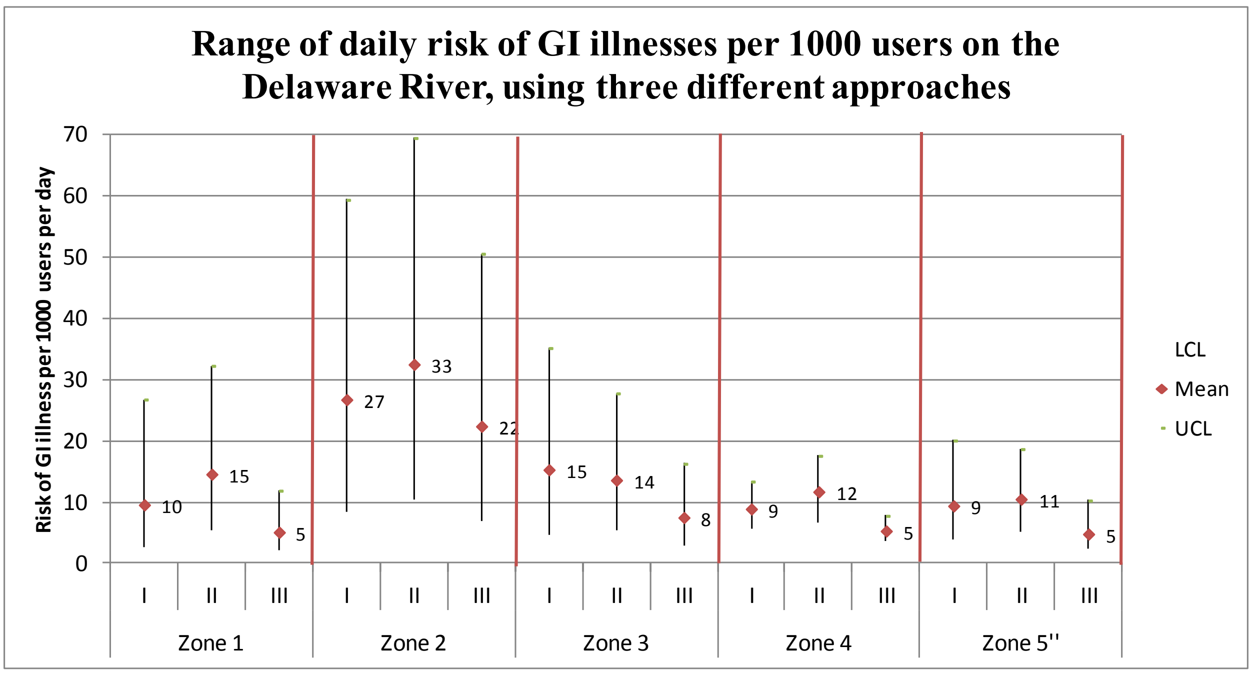

Figure 6-1 A comparison of 95% confidence interval range of GI illnesses obtained from 3 methods, due to recreation in the Delaware River under wet weather conditions.

The results obtained for approach 2 represents risk per 800 users, which excluded boating exposure. For this approach we separately calculated the risk estimates for boaters assuming 200 boaters to be exposed to the river water per day. The enterococci distribution obtained per site was averaged to obtain an envelope distribution of water quality for boaters crossing all the 5 sites. This analysis resulted in total 5.1 illnesses/200 boaters/day with a 95% confidence interval range of 2.9 to 8.3 cases/day.

Table 6-11 Average daily estimated cases of GI illnesses and contribution to total risk by each activity for Delaware River, based on approach 1, 2 and 3.

Risk of GI illnesses per day per zone	Zone 1			Zone 2			Zone 3			Zone 4			Zone 5		
	Per 1000 users (App.1)	Per 800 users (App. 2)	Per 1000 users (App.3)	Per 1000 users (App.1)	Per 800 users (App. 2)	Per 1000 users (App.3)	Per 1000 users (App.1)	Per 800 users (App. 2)	Per 1000 users (App.3)	Per 1000 users (App.1)	Per 800 users (App. 2)	Per 1000 users (App.3)	Per 1000 users (App.1)	Per 800 users (App. 2)	Per 1000 users (App.3)
Avg. Risk	9.57	8.57	5.09	26.75	26.47	22.39	15.26	7.97	7.57	8.87	5.65	5.33	9.38	4.51	4.83
Boating	*6.46*	-	*2.9*	*6.03*	-	*1.98*	*9.5*	-	*2.14*	*4.5*	-	*1.24*	*6.4*	-	*2.36*
Wad/Play	*0.33*	*0.9*	*0.3*	*4.28*	*5.4*	*4.35*	*0.27*	*0.37*	*0.27*	*1.21*	*1.58*	*1.22*	*0.17*	*0.25*	*0.17*
Jet/kayak	*1.59*	*4.5*	*0.68*	*0.43*	*0.68*	*0.15*	*0.39*	*0.53*	*0.1*	*0.39*	*0.5*	*0.1*	*0.52*	*0.8*	*0.05*
Fishing	*1*	*2.67*	*0.99*	*10.97*	*13.97*	*10.87*	*4.9*	*6.79*	*4.86*	*1.32*	*1.72*	*1.34*	*1.31*	*1.98*	*1.31*
Swimming	*0.19*	*0.51*	*0.19*	*5.03*	*6.4*	*5.03*	*0.2*	*0.28*	*0.2*	*1.44*	*1.86*	*1.44*	*0.58*	*0.88*	*0.57*
Duck tour													*0.39*	*0.59*	*0.39*

The overall risk estimate for recreation in the Delaware River was calculated as sum of estimates obtained per zone by each approach. Thus the risk of illness per 1000 users per day was obtained as 69.83, 58.27 and 45.21 based on approach 1, 2 and 3, respectively. It should be noted that under approach 1 the assumption of linearly relating the exposure durations of transient activities to the number of segments in each zone might have resulted in increased risk calculations for zone 3, 4 and 5. These zones had 6 to 8 segments as compared to 3-4 segments in other zones. Also with all the other factors being same between approach 1 and 3 except the transient activities duration, the 35% lower estimates with approach 3 indicates that the duration statistics can significantly affect the risk prediction and should be re-evaluated for more confident risk calculations.

6.4 Conclusion

The base line risk assessment was conducted to predict the overall risk potential of each observed activity along the Schuylkill and the Delaware River. Risk predictions were made per zone and the critical zones were identified where the estimates were significantly high. Collectively, on-shore fishing and boating were found as major activities leading to significant potential of health risk to users when exposed during wet weather discharges. For the Schuylkill River zone 3 – downstream of Bartram's Garden (~5.94 river miles) and for the Delaware River zone 2- between Pennypack Creek to Tacony bridge (~ 2.5 river miles) were identified as priority regions to be investigated further for more comprehensive data collection. As a caution, the public health officials can take measures to limit fishing near active CSO points in identified reaches of these rivers to avoid any adverse health effect from recreation. This finding is also very helpful in efficient planning of resources to collect more data in future for further analysis. Multiple sampling points to collect more water quality data and exposure data at these reaches can help us to develop a better framework for monitoring recreational health impacts in Philadelphia. The approach developed in this analysis is based on various assumptions but it successfully provided a base line method for expanding the sampled site data to the vast reach of the river and provided a useful tool for policy makers to make informed decisions.

7 CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE WORK

7.1 General Conclusions

In this research, the recreational water health hazards were estimated for the City of Philadelphia based on enterococci and *E. coli* data from the recreational seasons of 2007, 2010 and 2011. Ten water locations distributed across 4 water bodies were evaluated for assessing the exposure scenarios and to derive the indicator organism based risk estimates.

The first major objective of this research was to identify the current recreational use pattern at Philadelphia waters and to develop exposure statistics for each of the observed activity. Exposure factors for non-swimming activities were of main concern as these activities are the principal mode of exposures to surface water bodies in the City. In this effort, we successfully demonstrated the application of remote surveillance in collecting human exposure data and generated a user friendly archive of all the recorded activities per site for an organized reference in future. Our results obtained from camera surveillance when compared with in-person data showed a strong correlation (R^2 = 0.81 to 0.96) between the two survey techniques, indicating that the application of remote photography in collecting human exposure data was appropriate. As a main output we developed probability distributions for exposure durations of playing, wading and fishing at rivers and creeks; which can be of potential use for other studies where non-swimming activities are predominant. We also outlined a framework for future use of remote surveillance to collect human recreational-exposure data. This is a useful tool as currently there is no information in open literature for the technical specifications and scientific approach required to use this technique in order to collect human-exposure data.

The second major objective was to quantify the health risk associated with recreational exposure to surface water bodies during the period of wet weather discharges. For this purpose, we quantified the daily risk of HCGI illnesses for dry and wet weather conditions using probabilistic techniques and presented the results under two categories; risk per 1000 users and total risk per observed number of users, using enterococcus and *E. coli* as fecal indicator organisms. Exponential dose response models were developed for each indicator organism as an alternative to the 1986 US EPA's log-linear model. At all of the sites dry weather risk estimates were found significantly lower than the wet weather estimates, indicating increased health hazard to users after a heavy rainfall event. On comparing the wet weather risk estimates with the US EPA limit of less than 8 illnesses/1000 users/day, we identified 2 prominent sites- COB and AD, where under episodic events of poor water quality the risk exceeded the acceptable value. This does not necessarily indicate that these sites violated the criteria, as the acceptable level of 8 illnesses are based on geometric means of the water quality data recorded over time as oppose to the daily values which were used in the present study. But since the risk levels were found elevated, we recommend more data collection for better understanding of health hazard at these locations. In general, the uncertainty associated with the bacterial density distribution contributed the most to the total uncertainty of the risk estimates at creeks and Schuylkill River sites. However at Delaware River, both the water quality parameter and the exposure factors appeared equally important in influencing the risk distribution. Another important finding of our analysis is the identification of GI risk potential of wading and playing in water. The risk estimates were developed for all users, in proportion to the frequency of use, for each water location. The recreational activity with the highest exposure potential was swimming at creeks; however depending upon the users proportion and exposure duration, wading and playing in water resulted as most critical activities causing greatest number of affected users at creek locations.

The third and final objective of this work was to generate the overall risk estimates for the Philadelphia region of the Schuylkill and Delaware River based on the sampled site observations and to identify the critical reaches of these water ways that would need further investigations. For this purpose, we developed a base line approach to integrate our health risk from local observation stations to the full stretch of the river. Both the rivers were divided into consecutive zones and the risk per zone was projected based on enterococci counts measured at sampled locations. The Schuylkill River region downstream of Bartram's Garden and the Delaware River region between Pennypack Creek to Tacony Bridge were identified as priority regions to be investigated further for more comprehensive data collection. Across the range of activities observed along the rivers, which included boating, on and off shore fishing, jet skiing, kayaking, wading, playing, swimming and duck boat tours; the activity causing greatest contribution to the total number of affected users was on-shore fishing. To reduce the risk, the posting of "No Fishing" signs can be considered near active CSO points in the critical zones of these rivers.

The overall contribution of this dissertation includes the development of an exposure assessment tool to monitor and analyze human exposure to recreational waters, generation of stochastic exposure duration distribution models for non-swimming activities by size of the water body (small creeks and large rivers), identification of critical activities – wading, playing and on-shore fishing that presented greatest potential to cause GI illnesses to the users under current usage profiles and marking of the priority zones on the two large rivers that needs to be further investigated for more robust bacterial and exposure data collection.

7.2 Limitations and Recommendations for future work

This research has been successful in modeling recreational health risk profile of Philadelphia waters. However, there are a few limitations that need to be addressed to make our analysis and modeling more robust.

One of the major limitations in this study is the uncertainty associated with the transient activity duration estimates. Person hour of use by activity was estimated at each location on the Delaware River for boating and jet skiing instead of the actual length of time for which users were engaged in these activities. A future endeavor in this research could be to try contacting the private boat clubs and row houses on these water bodies to collect estimates for boating duration and daily counts of boaters. The camera observed boaters count could be validated with the survey counts obtained from the boat house managers. Furthermore, the sensitivity analysis presented in this research was one dimensional, which can be more detailed by conducting a 2D-Monte Carlo simulation. Such probabilistic assessment could provide a clear understanding of uncertainty combined with variability in assessing the collected dataset. Finally, our analysis considered only ingestion as the major route of exposure and gastrointestinal illnesses as the disease endpoint. However, dermal and fish consumption are other possible exposure pathways that can potentially lead to a wider range of illnesses including dermal, ear and eye infections. For a comprehensive risk assessment model these exposure routes should also be included to give an overall estimate of public health profile for recreational water exposures in the city. However, such a multi-level risk assessment approach was outside the scope of this study due to time constrains.

In addition to these limitations, we identified various areas where further research can promote a better understanding about the exposure patterns and the health risk profile of the Philadelphia recreational-water users. The areas of further research include the following –

1) Forecast of recreational use - The exposure model developed in this study predicted the usage spatially (along the length of the river); however with more exposure data the predictions can also be made temporally. Based on our survey, there is a huge scope of infrastructure advancements along the shoreline and thus the recreational demand is also expected to increase with these developments. Using the exposure model developed in this study, coupled with the information collected in the online surveys regarding the developments planned per river-segment; recreational demand in future can also be modeled. Most commonly used methods for generating quantitative forecast models associated to recreational exposure are either time based methods or regression analysis. In addition to this, forecasts based on population projections are also widely by many researchers (Archer 1987). A study conducted by Saunders and his team on recreation at the upper Savannah River basin, used regional participation rates, population projections and distance to recreation sites to predict total and peak season recreational demand (Saunders, Senter et al. 1981). Depending upon the forecasting period and required accuracy, a site-specific aggregate approach wherein population data are combined with individual site level data to project participation and use by activity can be planned in future to predict usage demand at the Schuylkill and the Delaware Rivers.
2) Pathogen based risk model – In addition to bacterial indicator microorganism, human viral and protozoan pathogens that are typically present in the feces have gained major attention in the scientific community because of their longer persistence than coliform

bacteria (Griffin, Gibson et al. 1999). In addition to this, many water- borne recreational illnesses are viral and since FIB do not always indicate the presence of viruses, the use of indicator bacteria to represent fecal pollution is often compromised (Wong, Kumar et al. 2009). Some of the potential pathogens reported to be present in recreational waters are *Giardia lamblia*, *Cryptosporidium parvum*, polioviruses, coxsackie A and B viruses, echoviruses, hepatitis A viruses, Norwalk viruses, Rotavirus, Norovirus and Adenovirus (Rose, Mullinax et al. 1987; Griffin, Donaldson et al. 2003; Sinclair, Jones et al. 2009). Many researchers have conducted simultaneous pathogen based risk projections along with the indicator organism based analysis to present the full spectrum of health risk associated with recreational waters (Donovan, Unice et al. 2008; Geosyntec 2008). Hence, future efforts can be made to perform water quality analysis for potential pathogenic protozoans and viruses for which EPA approved laboratory detection methods are available and the corresponding dose-response models are also developed. The overall risk estimates based on pathogens and indicator organism will make this analysis more robust and will provide a better risk management tool to the City officials.

3) <u>Comparative analysis with alternative indicator organism</u> – As mentioned previously, the membrane filtration method for *E. coli* and enterococci quantification is inadequate in providing the users with timely information regarding recreational health risk (Porter 2004). Also, there is a raised concern about the specificity of indicator organisms to human sources of pollution (McQuaig, Scott et al. 2006). The entry of indicator bacteria in significantly large amounts from stormwater drainage (Parker, McIntyre et al. 2010), impervious surface runoffs (Gaffield, Goo et al. 2003; Sauer, Vandewalle et al. 2011) and upstream agricultural lands (Layton, McKay et al. 2006) without having any human

component to it, raises the question about their use for representing the health risk associated with CSO discharges. These multiple pathways for entry of fecal pollution in the water systems results in addition of non-human sources of contamination such as contribution from urban wildlife, livestock, pets and agricultural animals (Newton, VandeWalle et al. 2011). Many researchers have investigated the application of alternative indicator organisms that have better host-specificity than traditional indicator organisms, such as *Lachnospiraceae* and *Bacteroidales* (Saunders, Kristiansen et al. 2009; Newton, VandeWalle et al. 2011; Sokolova, Åström et al. 2011). The Philadelphia Water Department is making massive investments to mitigate CSO discharges, particularly to reduce the excess risk of health hazard associated with these discharges. However, using traditional indicator organisms we might be over estimating the extant of pollution and thereby, the related health risk due to the reasons explained above. Hence, use of such human-specific fecal indicator organism in evaluating the impact of wet weather discharges should be tried in future to have improved assessment of health risk association with discharges from CSOs. However, in absence of any dose-response model their application in modeling health risk is not possible under current situation. With increasing awareness it might be possible in future to have a good source of data to develop dose-response models for these alternative organisms and hence use them for predicting associated health risk for recreational exposures.

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Appendix- A

Table A.1: List of contacts for survey response

River Segment	Contact	Affiliation
Both rivers	Jeffery Knowles	Project Manager, Southeast office, PEC
Both rivers	Jessica Anderson	Ex-PEC, Coordinated Kayaking activities for Delaware river
Schuylkill Banks-Lower SRT	Joseph Syrnick	President & CEO Schuylkill River Development Corporation
Schuylkill Banks-Lower SRT	Danielle Gray	Communications Manager Schuylkill River Development Corporation
Upper SRT Del. – Parks ONLY	Rob Armstrong-	Preservation & Capital Projects Manager Philadelphia Parks & Recreation
Upper SRT – Manayunk	Kay Sykora	Project Director Schuylkill Project/New Manayunk Corporation
PH and Pulaski Parks	Barbara McCabe	Director Office of Volunteers and Stewardship Philadelphia Parks & Recreation
Upper Delaware	Thomas Branigan	Executive Director Delaware River City Corporation
Middle Delaware	Sarah Thorp	Master Planning Manager Delaware River Waterfront Corporation

The Schuylkill River Survey forms-

1) Link for City line to Falls Bridge survey form (7 segments)-
https://adobeformscentral.com/?f=S6C2oGJhWJE-jKLEqhj3bA

2) Link for Falls Bridge to Fairmount Dam survey form (13 segments) -
https://adobeformscentral.com/?f=WwQBTGaKdCJCOs5LA1J1ZA

3) Link for Schuylkill Banks survey form (13 segments)-
https://adobeformscentral.com/?f=eiZxw27S1r2twYuVgEaBtQ

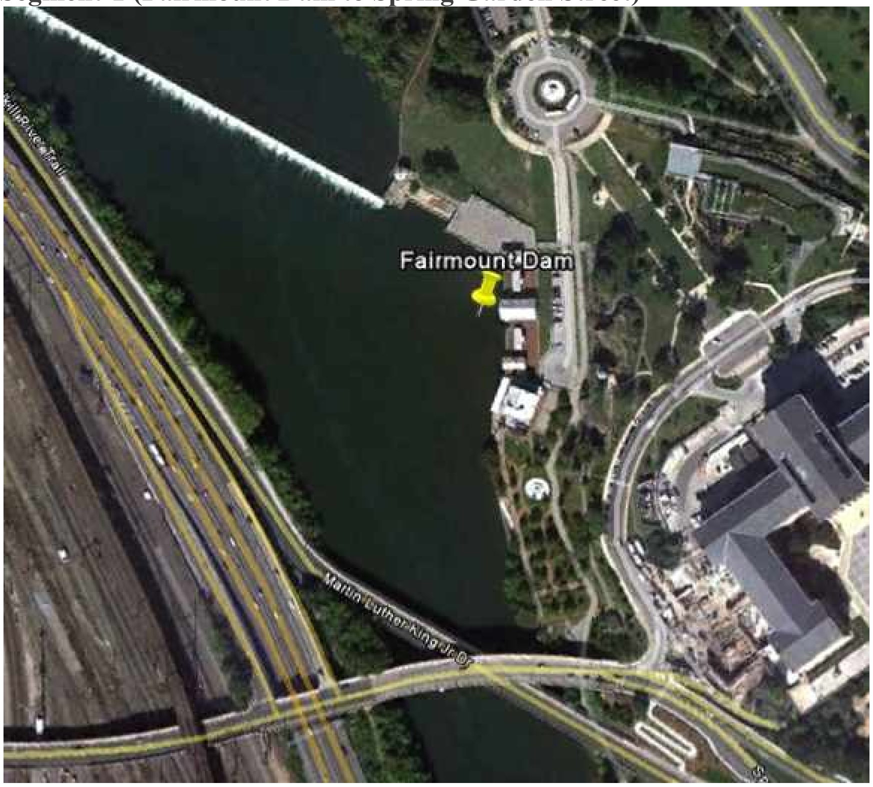
The Delaware River Survey forms-

1) Link for City line to Allegheny Ave form (13 segments)-
https://adobeformscentral.com/?f=7NG8tV2miyCWIoQICh9DSw

2) Link for Allegheny Ave to Oregon Ave form (14 segments)
https://adobeformscentral.com/?f=nc7BVwejzGi837axVKi%2AlA

Appendix- B

Segment-1 (Fairmount Dam to Spring Garden Street)



1.1 **What is the possibility of observing following activities in Segment 1 on a given day of the recreational season?***

	Less than 1 in 10	Almost 2 in 10	Almost 5 in 10	Almost 8 in 10	More than 9 in 10
Boating	○	○	○	○	○
Boat Fishing	○	○	○	○	○
Fishing on East Side	○	○	○	○	○
Fishing on West Side	○	○	○	○	○
Jet Skiing	○	○	○	○	○
Kayaking	○	○	○	○	○

1.2 Is it likely to have a new access point on this segment in next 5 years?

	Not Likely	Moderately Likely	Strongly Likely
Fishing Pier			
Boat Dock			
Jet Ski Dock			
Kayak Dock			

- If you chose likely on any category above, please provide details
- Fishing Pier
- Boat Dock
- Jet Ski Dock
- Kayak Dock

Appendix- C

Below are listed the segment boundaries for online-survey forms; the segments monitored by cameras are highlighted as Bold.

Upper SRT Segments – (4.88 miles)

1. **City line at Montgomery County line to Flat Rock Dam (~1.4 miles)**
2. Flat Rock Dam to Leverington Ave. (~1.12 miles)
3. Leverington Ave.to Green Ln. (~0.3 miles)
4. Green ln to Lock St. (~0.44 miles)
5. Lock St. to Wissahickon Creek (~0.95miles)
6. Wissahickon Creek to City Ave. (~0.20miles)
7. City Ave to Falls Bridge (~0.47 miles)

Lower SRT Segments 9.6miles) –

1. **Fairmount Dam to Spring Garden st. (~0.5miles)**
2. Spring garden to Vine St. (~0.8miles)
3. Vine St. to Walnut St. (~0.3 miles)
4. Walnut St. to South St. (~0.44 miles)
5. South St. to University Ave. (~0.95miles)
6. University Ave. to Wharton Ave. (~0.20miles)
7. Wharton Ave. to N. of Bartrams Garden (~0.47 miles)
8. **Bartram's Garden (~0.53 miles)**
9. S. of Bartrams Garden to Passyunk (1.01 miles)
10. Passyunk to S. 67th St. (1.03 miles)
11. S. 67th St. to Penrose Ave. (0.91 miles)
12. Penrose Ave. to I-95 (0.81 miles)
13. I 95 to Fort Mifflin (1.65 miles)

Upper Delaware Section:-

1. City line at Poquessing Cr. to Linden Ave (~ 0.9 miles)
2. **Linden Ave. to Edge of intake (Pleasant Hill Park~0.3 miles)**
3. Edge of intake to Upper end of Pennypack Park (~0.7 miles)
4. Pennypack creek to middle of PP park (~0.42 miles)
5. **Middle of PP park to Rhawn St. (~0.42 miles)**
6. Rhawn St. to Princeton Ave. (~0.9 miles)
7. Princeton Ave. to Tacony Bridge (~0.75 miles)
8. Tacony Bridge to Robbins St (S. of Lardners Park)Brill St. (~0.18 miles)
9. Robbins St to N. of Arsenal Boat launch (~0.51 miles)
10. **N. of Arsenal Boat launch to Old Frankford Cr. (~0.25 miles)**
11. Old Frankford Cr. to Frankford Cr (~1.5 miles)
12. Frankford Cr to Pulaski Park (~ 1.5 miles)

13. Pulaski Park to Allegheny Ave. (~0.03miles)

<u>Middle Delaware Section:-</u>

1. Allegheny Ave to Lehigh Ave. (~ 0.7 miles)
2. Lehigh Ave to Cumberland St. (~0.38 miles)
3. Cumberland St. to Berks St. (~0.5 miles)
4. Berk St. to N. of Penntreaty Park (~0.25 miles)
5. **N. of Penn Treaty Park to Marlborough St. - Penn Treaty Park (~ 0.14 miles)**
6. Marlborough St. to S of Noble St. (Spring Garden Park) ~0.50 miles
7. S. of Spring Garden Park to Ben. Frank. Bridge (~0.3 miles)
8. Ben. bridge to Market St.(~ 0.3 miles)
9. Market St. to Walnut St. (~0.25 miles)
10. Walnut St. to Lombard St. (~ 0.35 miles)
11. Lombard St. to Washington Ave. (~0.6 miles)
12. Washington Ave to Tasker St. (~ 0.43 miles)
13. Tasker St. to Mifflin St (~0.2 miles)
14. Mifflin St. to Pattison Ave (~1.5 miles)

Appendix- D

Sample calculation:-

Table D.1: Summary of the percent of respondent agreeing with the same probability of observing any given activity in Fairmount Dam segment on Schuylkill River survey.

Percent of Survey Responses at Fairmount Dam segment on the Schuylkill River (Total 3 responses)

	Prob. of observing following activities on a given day					Survey		Cam. Based no.	Predicted Users =
Activities	<1/10	2/10	5/10	8/10	>9/10	ranking	Survey pdf	of users/day	cam. total*survey pdf
Boating		66.70%	33.30%			3.0	= 3.0/13.7=0.22	9	45*0.22 = 10
Boat fishing		100%				2.0	0.15	6	45*0.15 = 7
Fishing on east side			66.70%		33.30%	4.2	0.31	12	45* 0.31 = 14
Fishing on west side	66.70%		33.30%						
Jet skiing	33.30%	66.70%				1.5	0.11	3	45* 0.11 = 5
Kayaking		66.70%	33.30%			3.0	0.22	15	45 *0.22 = 10
					Sum=	13.7	Sum=	45	

The calculation approach for survey ranking for each activity in the given segment is explained below by the sample calculation for boating activity –

1) 66.7% respondents agreed for 2/10 probability,
2) 33.3% respondents agreed for 5/10 probability, for observing boating in the given segment. Thus the corresponding survey based ranking for boating is given by=
    a. $\text{Ranking} = \frac{66.7*2}{100} + \frac{33.3*5}{100} = 3.0$

Vita

Neha Sunger was born in Mumbai, Maharashtra, India. She moved to Jaipur, Rajasthan with her parents when she was 5 years old. Upon completion of her high school education from Brightlands School, Jaipur, she earned a bachelor's degree in civil engineering in June, 2003 from the MBM Engineering College, Jodhpur, Rajasthan. She then worked as Assistant Engineer in pile foundation and ground engineering division of Simplex Ltd., New Delhi, India. In July, 2004, she undertook graduate studies at the Indian Institute of Technology, Kanpur (IIT-K) and earned her master's degree in environmental engineering and management. In her masters work she developed a reactor configuration which facilitated on-site groundwater treatment for nitrate removal through hydrogenotrophic denitrification, assisted by hydrogen generated from anoxic corrosion of metallic iron. In July, 2006, she joined one of the India's leading energy and environment company, Thermax, Ltd., Pune, as Senior Executive Engineer. At Thermax Ltd., she worked on design of water and wastewater treatment schemes and project management. She started her doctoral studies at Drexel University, PA, in July, 2008, in environmental engineering and received the research grant from Philadelphia Water Department (PWD) to conduct microbial health risk assessment for recreational water-users in Philadelphia.

RECENT PUBLICATIONS

- **Sunger N.**, Teske S.S., Nappier S., and Haas C.N. Recreational use assessment of water-based activities, using time-lapse construction cameras. *J Expos Sci Environ Epidemiol* 2012: 22(3): 281-290.
- **Sunger, N.,** Nair, C., Solon, I., Kundu, A., Nappier, S. and Taft, S. "Human Bocavirus: A Risk Assessment" research article at *Center for Advancing Microbial Risk Assessment*-2009.
- **Sunger, N.,** and Haas, N. "Recreational health risk assessment for secondary contact activities in Philadelphia waters" *under review at Journal of Water and Health..*
- **Sunger, N.**, Prasad, B., Lennon, E. and Haas, C. N. "A risk model for inhaled toxins associated with spores of Stachybotrys Chartarum", *under preparation.*

CONFERENCE PROCEEDINGS

- <u>Asset management</u>: Muir, R., **Sunger, N.,** Wittke, R. and Gurian, P. "Managing Catastrophic Risk to Istanbul's Housing Infrastructure" *International Conference on Bridge Maintenance, Safety and Management (IABMAS,2010).*

- Exposure assessment: **Sunger,N** and Haas, C.N. " Recreational Water Exposure Assessment using Time-lapse photography", *poster presentation*, Salt Lake City, Utah, Society for Risk Analysis (SRA) Annual Meeting, Dec. 2010.
- Source water protection: **Sunger,N** and Haas, C.N. " Recreational usage impact on source water quality", *poster presentation*, Philadelphia, PA, American Water Works association (AWWA) annual conference, June 2010.
- Risk assessment: **Sunger,N** and Haas, C.N. "Quantitative Microbial Risk Assessment for Gastrointestinal illnesses associated with recreational water exposure, in the Philadelphia Area", *poster presentation*, San Francisco, California, Society for Risk Analysis (SRA) Annual Meeting, Dec. 2012.